Human Health Risk Assessment for the Donlin Gold Project, Alaska

June 2017

ERM Project # 0401326

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EXECUTIVE SUMMARY

This HHRA evaluated the potential addition of metal constituents to the environment in areas where subsistence populations live and harvest wild foods near the proposed Core Operating Area of the Donlin Gold project (Project). The approach of the HHRA was to evaluate incremental risk; that is, what is the added human health exposure and associated incremental risk due to predicted future concentrations of certain metals in soil, sediment, surface water, wildlife and fish that may be harvested and eaten near the Project as well as from inhalation associated with air emissions. Incremental risk from antimony, arsenic, and mercury (constituents of potential concern, or COPCs) were quantified in the HHRA. A health-protective approach was taken to quantify incremental exposure to these constituents, in that conservative exposure assumptions were used to calculate potential incremental risk to subsistence populations that may be hunting, gathering, or living in the area around the proposed Core Operating Area.

A deterministic-based computation of non-cancer and cancer risks was completed following Alaska Department of Environmental Conservation (ADEC) and United States Environmental Protection Agency (USEPA) guidelines for HHRAs. The HHRA integrated the results of the environmental media baseline studies, human receptor characteristics, subsistence information, and agency-recommended toxicity values to estimate non-cancer and cancer risks.

CONCEPTUAL MODEL

A conceptual site model (CSM) was developed to describe the sources, transport mechanisms, exposure pathways, and media that may result in human exposure to COPCs. Both background sources and Project-related emissions contribute to total environmental exposure. The primary sources of Project-related constituents would be fugitive and point source air emissions. Surface water that comes in contact with mining infrastructure ("contact water") will be treated prior to discharge to levels at or below ADEC surface water quality criteria, which are protective of human health. Groundwater around the Core Operating Area is not a source of drinking water.

The HHRA considered exposures within a 20 mile radius outside the Core Operating Area, which represents the spatial extent of the mercury air deposition model (Environ 2015). This spatial boundary is referred to as the "study area". The HHRA calculated risks at the end of mine life (year 27), because the increase in constituent concentrations in the environment due to emissions from the proposed Project is expected to be highest at that time. After operations cease, Project emissions will diminish, and accordingly, evaluating potential exposure as of the end of operations represents a reasonable temporal boundary for maximum potential risk related to releases from the proposed Project.

Exposure pathways evaluated in the HHRA included inhalation of airborne constituents, incidental ingestion of constituents bound to soil, and ingestion of constituents associated with biota (fish, plants, and wildlife) that may be harvested in the study area. Because consideration of all individual species or wildlife trophic components of an ecological system is not practical or necessary in order to quantify incremental risk, representative species were evaluated in the

HHRA. Based on a review of the subsistence food items reported to be harvested by the Upper Kuskokwim group, and considering physiological factors of different species, the HHRA evaluated the incremental risk from consumption of northern pike (representing resident fish consumption), beaver (representing small mammal consumption), mallard ducks (representing wild bird consumption), and blueberries and cranberries (representing wild berry consumption).

The HHRA evaluated receptor populations that would have the highest exposure potential in the study area, which would be people who live in or near the Project and engage in subsistence activities such as harvesting, hunting, and fishing in the study area. Crooked Creek, located south-southeast of the proposed Project, and Georgetown, located east-southeast of the proposed Project, are the only villages in the study area. Based on Brown et al. (2012) and other studies cited in the Draft Environmental Impact Statement (DEIS; [USACE 2015]), the information on subsistence harvest patterns establishes that people living closest to the Project do not limit or concentrate their subsistence activities to the study area. Regional subsistence harvesting patterns shown in Brown et al. (2012) indicate that harvesting occurs predominantly outside of the study area. In order to be health-protective, a generalized subsistence population was conceptualized to evaluate exposures in the HHRA. This generalized population was assumed to live in the village of Crooked Creek (thus exposed to air and soil concentrations year around) and to hunt, fish, and gather only within the study area for beaver, mallard ducks, berries, and northern pike.

Quantities harvested and consumed were based on data presented for the Upper Kuskokwim subregion group (Brown et al. 2012), as this group lives nearest and most frequently harvests nearest to the proposed Project. Both the generalized subsistence population and assumptions about levels of consumption resulted in an overestimate of the exposure frequency and duration in the study area. This is a conservative analytical approach that ensures that potential effects on human health due to exposures via inhalation, as well as consumption of wild food collected within the study area, are not underestimated.

EXPOSURE ASSESSMENT

The calculation of chemical intake, or dose, requires estimation of exposure point concentrations (EPCs), and estimation of intake rates. Future (and some baseline) EPCs were estimated based on models rather than empirical data. To estimate future EPCs, the potential for future mercury methylation was evaluated, and bioaccumulation factors (BAFs) developed to understand potential future mercury bioavailability and constituent uptake into biota.

Methylation Rates

Atmospheric emissions from the Project have the potential to increase mercury loadings to the environment. Transformation of deposited mercury into methylmercury (MeHg) is of particular concern because: (a) it is the most toxic form of mercury and (b) it bioaccumulates in food chains to a greater extent than other, inorganic forms of mercury. The rate of conversion of total mercury (THg) to MeHg is the methylation rate, or %MeHg. The %MeHg is dependent on complex environmental factors. Numerous studies (e.g., USEPA 1997, Environment and

Climate Change Canada 2016, Frohne et al. 2012, Houben et al. 2016, Scudder et al. 2009, Ullrich et al. 2001) have attempted to quantify these factors, sometimes with conflicting results and rarely with quantifiable relationships. The primary and secondary mechanisms that affect %MeHg are identified as the availability of mercury, followed by oxygen, sulfate, organic carbon, pH and temperature. Other tertiary mechanisms such as photomethylation reactions may occur on a localized basis. Of the factors that have been identified as potentially affecting %MeHg, the Project is expected to change the THg and sulfate content through aerial deposition. These factors were incorporated into future estimates of MeHg in soil and the aquatic environment.

Future %MeHg was estimated using simple models based on empirical data. The Project is estimated to increase mercury deposition over baseline in the nearest watersheds, decreasing with distance from the Project. Atmospheric mercury that would be deposited would consist of gaseous mercury (Hg(0)), oxidized mercury (Hg2+), and particulate mercury (Hg(p)), with the majority in the particulate form. A minor component (approximately 1 to 2 percent) would be deposited as oxidized mercury, a form that is more likely to be methylated. Environ (2015) estimates that the deposition rate of Hg2+ from the Project sources will be approximately 2% of total Project deposition. Though Hg2+ can easily be methylated, the rate of methylation for the newly deposited Hg2+ is not known for the study area. As different forms of Hg may have different methylation rates, and "new" deposited Hg may be methylated more rapidly, instantaneous methylation of the newly deposited Hg2+ was assumed for purposes of the HHRA. Of the deposited mercury, 2% was assumed to be Hg2+, and 98% was assumed to be Hg(p). The USEPA (2005a) equation to calculate soil concentration based on deposition rate was used to calculate the future soil Hg2+ and non-Hg2+ concentrations from Project-related atmospheric deposition. All deposited Hg2+ was assumed to be converted to MeHg, and deposited non-Hg2+ is assumed to undergo methylation at a rate of 1%, twice the median of Project baseline data, but reflecting the mean of paired soil samples collected in 2014.

Increases in sulfate concentrations in natural systems with low concentrations of sulfate (as within the study area) will increase concentrations of MeHg. The reviewed literature indicates that the factor increase in MeHg is 0.5 to 1.0 times the factor increase in sulfate, or in other words, the increase in %MeHg would be of a similar magnitude to the increase in sulfate concentrations in systems where sulfate is initially at low concentrations. The estimated sulfur dioxide (SO_2) emissions (including fugitive emissions) from the proposed Project are predicted to increase atmospheric sulfate concentrations by 0.04 $\mu g/m^3$ at the Core Operating Area boundary, which equates to a 3% increase in atmospheric SO_2. A highly conservative assumption would be that the SO_2 concentration throughout the study area increased by this same amount, and that all the SO_2 produced was deposited locally as aqueous sulfate, increasing local annual load by 3%. Applying this conservative assumption in calculating %MeHg of existing soil THg, and newly deposited non-Hg2+, increases the initial estimated methylation rate of 1% to a rate of 1.03%.

Compounding these factors would result in soil THg concentrations increasing by a factor of 1.01, and MeHg increasing by a factor of 1.05. Mercury concentrations and %MeHg values are similar between soil and sediment datasets, implying similar processes and rates for both

systems. Thus, to estimate future concentrations in aquatic systems, the same factor increase computed for soil systems is used to derive THg and MeHg in sediments. Changes in sediment mercury content are then proportionally reflected in surface water THg and MeHg concentrations. Using this conservative analysis, future surface water THg concentrations are predicted to be below the EPA-approved Alaska state water quality criterion of 12 ng/L.

Bioaccumulation Factors

Because future concentrations of the constituents in subsistence food items cannot be measured, they were estimated by developing BAFs. Project-specific baseline data, other regional studies, and the general literature were reviewed to determine appropriate BAFs for the HHRA. Unlike arsenic or antimony, mercury can bioconcentrate in biota, and the bioconcentration factor is much higher for organic forms of mercury compared to inorganic forms. Therefore, differences in the bioavailability between inorganic and organic mercury forms were considered in deriving BAFs. BAFs were developed this way to account for potential changes in bioavailable mercury in the soil. MeHg is readily bioavailable and taken up by biota whereas the bioavailability of THg is more variable. Therefore, deriving MeHg to THg BAFs allows for some accountability of increasing future concentrations of MeHg in soil.

Compared to other literature or regional information, the Project-specific BAFs are high, suggesting BAFs may be overestimated. Reasons for the higher Project-specific BAFs are not known, but may in part be due to the nature of the sampling events during baseline collection, which sought to characterize the more mineralized areas of the study area. These more mineralized areas are not representative of the study area as a whole. However, the bias represents a conservative assumption.

Exposure Point Concentrations

An EPC is the concentration of a constituent in an exposure medium at the location where a receptor may contact that medium, and is representative of the time period over which exposure may occur.

For media baseline EPCs with empirical data, USEPA guidance and the associated software program proUCL v5.1 was used to calculate EPCs. Media that did not have empirical data included small mammals, waterfowl, and arsenic and antimony concentrations in berries; EPCs were estimated for these media using BAFs. Air concentrations were based on baseline analyses completed by Environ (2015). Summary statistics generated for baseline EPCs included both upper-bound and mean estimates. However, future EPCs are based on modeling estimates, which generate mean estimates. Therefore, mean baseline EPCs were used in computations of baseline risk.

Future air EPCs were estimated from modelling completed by Environ (2015) and Air Sciences (2017). Future soil EPCs were estimated following USEPA methods for calculating constituent concentrations in soil due to atmospheric dust deposition (USEPA 2005a), with subsequent increases in the proportions of THg and MeHg as described above. Constituent dustfall onto aquatic systems is assumed to be initially bound into stream sediments, then to become available for uptake into fish through the sediment. Therefore, dust deposition into sediments

was also estimated using USEPA (2005a). As described above, changes in surface water mercury would then be proportional to future soil and sediment concentrations, as surface water concentrations would reflect inputs by sediment dissolution into surface water and/or soil runoff.

The potential solubility of arsenic and antimony into surface water from sediments or soil runoff was assumed to be negligible, particularly after factoring the contribution of treated water from the water treatment plant, which will be at concentrations less than 0.005 mg/L (below water quality criteria). Therefore, future concentrations of arsenic and antimony in surface water were assumed to be unchanged from baseline concentrations.

Exposure Factors

Exposure factors define the magnitude, frequency, and duration of exposure for the populations and pathways selected for quantitative evaluation. Exposure factors are combined with EPCs to calculate dose. For purposes of this HHRA, exposure factors were selected based on a "reasonable maximum exposure" scenario that combines upper-bound and average values that reflect exposures somewhere between the 90th and 98th percentile of the range of possible exposures that reasonably can be expected to occur for a given population (USEPA 1989).

Exposure equations from the *Risk Assessment Guidance for Superfund, Human Health Evaluation Manual, Part A* (USEPA 1989) were followed in the HHRA. Consumption (i.e., ingestion) rates of food items were based on Brown et al. (2012), a subsistence study that reported harvesting data for the Upper Kuskokwim River subsistence population. As stated previously, the HHRA evaluated a generalized subsistence population conceptualized to evaluate exposures in the HHRA. This generalized population was assumed to live in the village of Crooked Creek (thus exposed to air and soil concentrations year around) and to hunt, fish, and gather only within the study area for beaver, mallard ducks, berries, and northern pike. Consumption rates of subsistence foods were based on per capita harvest rates (kg/year) for the Upper Kuskokwim Group reported by Brown et al. (2012). Because no information is available to estimate the proportion of harvested food that was subsequently consumed, it is assumed, for the purposes of the HHRA, that all harvested food items are consumed. The use of harvest rates to estimate consumption rates is an overestimate; the study indicates that people do not consume all of each animal harvested (e.g., often just the fish fillet is consumed, not the entire fish), and further, people may not necessarily consume the total amount of subsistence food harvested (e.g., primary subsistence providers frequently share with others who may reside in the area or in other distant communities). However, by utilizing the harvested amount to estimate intake, this HHRA provides a "health-protective" estimate of potential exposures from this pathway: if no incremental risk is indicated by this approach, then groups ingesting only a portion of harvested foods are unlikely to be at risk.

For other intake rates and other exposure factors (e.g., body weights and averaging times), ADEC (2015) exposure factors for subsistence receptors, or, where not available, USEPA (2014) exposure factors for residential receptors were used to develop exposure rates.

TOXICITY ASSESSMENT

Exposure to constituents can result in cancer and/or non-cancer effects, which are characterized separately. The toxicity hierarchy to select toxicity values for the HHRA reflects guidance from USEPA (2003) and ADEC (2015). This system prioritizes toxicity values as the following:

- Tier 1: USEPA's Integrated Risk Information System (IRIS).
- Tier 2: USEPA's Provisional Peer Reviewed Toxicity Values (PPRTVs).
- Tier 3: Other resources as needed, such as the Agency for Toxic Substances and Disease Registry's Minimal Risk Levels (MRLs), or USEPA's Health Effects Assessment Summary Tables (HEAST) values.

USEPA's IRIS system is the source for the toxicity values of all constituents evaluated in the HHRA.

RISK RESULTS

Non-cancer risks were estimated by calculating hazard quotients (HQs), which were then summed across all exposure pathways and constituents to estimate a hazard index (HI) for each receptor. Additionally, the potential for carcinogenic effects of arsenic was evaluated by estimating the probability of developing cancer over a lifetime. The interpretation of HQs or HIs is typically that exposures at or below the reference level (i.e., HQ=1) are unlikely to be associated with adverse health effects, while exposures above the reference level increase the potential for adverse effects.

Conservative assumptions were incorporated into this HHRA to prevent Type II errors, which are the elimination of a constituent, area, or activity from further consideration when, in fact, there should be a concern (i.e., false-negative conclusion). In the risk assessment process, uncertainties are handled conservatively (i.e., health-protective choices are preferentially made). This strategy is more likely to produce false-positive errors than false-negative errors. Particularly for this HHRA, a number of conservative estimates were made to assure that risk predictions erred on the side of Type I errors (i.e., false-positive conclusion) rather than Type II.

Non-cancer risk estimates were all at or less than 1 for both baseline and future risks. Even with the conservative assumptions of the HHRA, HQs and HIs indicate that non-cancer effects are highly unlikely. Baseline risk estimates are consistent with a mercury hair study completed by the Alaska Department of Health and Social Services (ADHSS). ADHSS conducted MeHg testing of hair samples from pregnant women in selected communities including the Upper Kuskokwim River region communities. Every study participant had a hair mercury level that was below the Agency for Toxic Substances and Disease Registry (ATSDR) No Observed Adverse Effect Level (15.3 ppm), as well as the Environmental Public Health Program cut-off for follow-up (5 ppm) (ADHSS 2013, as cited in the DEIS), also indicating exposure to mercury in the region is not at unacceptable levels.

In the case of cancer risk estimates for arsenic (the only carcinogenic COPC), both baseline and future risk estimates are $5x10^{-5}$, which is within the USEPA risk management range of $1x10^{-4}$ to

$1x10^{-6}$. Future estimates of cancer risks changed from baseline results by less than $1x10^{-6}$, indicating essentially no unacceptable change in risk.

Arsenic is a naturally occurring metal often found at concentrations above "background", as referred to in regulatory cleanup levels in 18 AAC 75.341. Baseline concentrations of arsenic in the study area are naturally occurring and consistent with regional, state and nationally published studies of naturally occurring arsenic in soil (Gough et al. 1988). Due to the prevalence of naturally occurring arsenic in Alaskan soil, ADEC recommends that cumulative risk calculations do not include risk contributions from naturally occurring arsenic sources (ADEC 2009). Statistical comparisons of significance between baseline and future arsenic concentrations cannot be made because only a single future soil arsenic concentration was estimated. However, if baseline cancer risk estimates were subtracted from future cancer risk estimates, the resulting cancer risk for adults would be $5x10^{-7}$, well below acceptable risk thresholds.

Further, cancer slope factors were used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to arsenic. Upper-bound estimates conservatively exaggerate the risk to ensure that the risk is not underestimated if the underlying model is incorrect. The arsenic ingestion slope factor is based on one affected population in Taiwan concerning non-fatal skin cancer incidence, age, and level of exposure to arsenic via drinking water rather than food (USEPA 2017). The predominant form of arsenic in the drinking water was in an inorganic form (arsenic trioxide), which is a highly toxic form of arsenic and not the predominant form of arsenic that occurs in the study area. The confidence in the oral slope factor is considered to be low overall. Studies that show the strongest link between ingestion of arsenic and cancer involve ingestion of inorganic arsenic at elevated levels in drinking water (Naujokas et al. 2013). Arsenic in soil is less bioavailable, and an adjustment is made to the soil exposure concentration accordingly. There are additional uncertainties estimating cancer risk from arsenic because the mechanism of action in causing human cancers is not known, and studies on arsenic mutagenicity are inconclusive (USEPA 2017). However, safety factors included in the slope factor provide a conservative estimate of risk.

The low confidence in the arsenic slope factor, and the safety factors applied to the slope factor overall, may lead to an overestimate of potential cancer risk due to arsenic in the current environment, in addition to exposure assumptions that overestimate the amount of arsenic ingested by receptors. Further, the incremental changes in cancer risk due to future estimated arsenic exposures are very small. Given that the overestimated potential current and future risks are within the risk management range and are related to naturally occurring levels of arsenic, an unacceptable increase in cancer risk due to site-related arsenic exposure is unlikely.

Risks Due to Fish Consumption

This HHRA evaluated risks due to northern pike fish consumption. Resident fish were incorporated into the HHRA because of their relatively greater exposure period compared to

migratory fish. Northern pike, a large, relatively long-lived top predator that is found in the Crooked Creek watershed, was used to estimate fish EPCs and assess risk associated with subsistence fishing. Top predators, such as northern pike, also bioaccumulate mercury to a greater degree than lower-trophic level species. Tissue burdens for northern pike were estimated by applying trophic transfer factors to sculpin data collected for the Project. Estimates for northern pike indicate that both baseline and future HQs are at or less than 1, indicating that noncancer effects are unlikely from northern pike consumption. As noted in the previous section, the estimated incremental cancer risk increase is less than $1x10^{-6}$.

Estimates of future northern pike concentrations are also lower than State of Alaska (2016) fish consumption advisories for northern pike harvested from the Kuskokwim River in the vicinity of the Project. Therefore, subsistence populations currently consume northern pike at rates well below the northern pike consumption advisory level for the State of Alaska, and consumption would be expected to remain below the advisory level in the future.

As reflected in the DEIS comments, there is public concern regarding constituents in salmon species, which are a primary subsistence food in the region. Salmon species are present within the study area, including coho salmon. However, within the study area, only juvenile coho (and other species of) salmon reside within the Crooked Creek drainage system for extended periods. Adults, rather than juvenile species, are harvested for consumption. Juveniles migrate out of the freshwater system at an early age, spending the majority of their lifetimes in marine environments. These migratory fish gain the bulk of their body mass over a much larger exposure area (marine waters) that is entirely outside the study area. Because the exposure conditions of salmon are largely unrelated to the study area, the HHRA did not quantify risk due to salmon ingestion.

SUMMARY CONCLUSION

This HHRA demonstrates that the small increases in constituent concentrations estimated to occur outside of the Core Operating Area due to Project-related activities will not result in unacceptable risks to human populations who would have the highest exposure. Based on the exposures to the conservatively defined receptors analyzed in this health-protective HHRA, other human populations, such as residents in the region, would not be expected to be exposed to unacceptable risk due to exposure to Project-related concentrations of mercury, arsenic, or antimony.

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ACRONYMS AND ABBREVIATIONS

µg .....Microgram
%.....Percent
ADD.....Average daily dose
ADEC .....Alaska Department of Environmental Conservation
ADHSS .....Alaska Department of Health and Social Services
ATSDR.....Agency for Toxic Substances and Disease Registry
As .....Arsenic
BAF .....Bioaccumulation Factor
BLM .....Bureau of Land Management
BW .....Body weight
cm.....Centimeter
COPC.....Constituent of potential concern
CR .....Cancer risk
CSM .....Conceptual site model
D-MCM .....Dynamic Mercury Cycling Model
DEIS.....Draft Environmental Impact Statement
DL .....Detection limit
DOC.....Dissolved organic carbon
DOM.....Dissolved organic matter
Donlin.....Donlin Gold, LLC
dw .....Dry weight
EC.....Exposure concentration
EFSA .....European Food Safety Authority
EPC .....Exposure point concentration
FCM .....Food chain multiplier
g.....Gram
Hg .....Mercury
Hg(0).....Elemental mercury
Hg(2+) .....Oxidized/divalent mercury
Hg(p) .....Particulate mercury
Hg-S.....Mercury-sulfur complexes
HHRA.....Human Health Risk Assessment
HI .....Hazard index
HQ .....Hazard quotient
IR.....Ingestion rate
IRIS .....Integrated Risk Information System
kg .....Kilogram
L .....Liter
LADD .....Lifetime average daily dose
m .....Meter
m^2 .....Square meter

m^3 .....Cubic meter
MDN.....Mercury Deposition Network
MeHg.....Methylmercury
mg .....Milligram
n.....Sample size
ng .....Nanogram
NOAEL .....No-observed-adverse-effect-level
$PM_{2.5}$.....Particulate matter less than 2.5 microns
ppm.....Parts per million
Project.....the Donlin Gold Project
PSD .....Prevention of Significant Deterioration
RBA.....Relative bioavailability
redox.....Oxidation-reduction
RfC .....Reference concentration
RfD .....Reference dose
RI.....Remedial Investigation
Sb.....Antimony
SERAFM.....Spreadsheet-based Ecological Risk Assessment for the Fate of Mercury
SF.....Slope factor
S2- .....Free sulfide
SO_2 .....Sulfur dioxide
SRB.....Sulfate reducing bacteria
THg.....Total mercury
URF.....Unit risk factor
USACE .....United States Army Corps of Engineers
USEPA.....United States Environmental Protection Agency
USGS.....United States Geological Survey
ww .....Wet weight

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1. INTRODUCTION

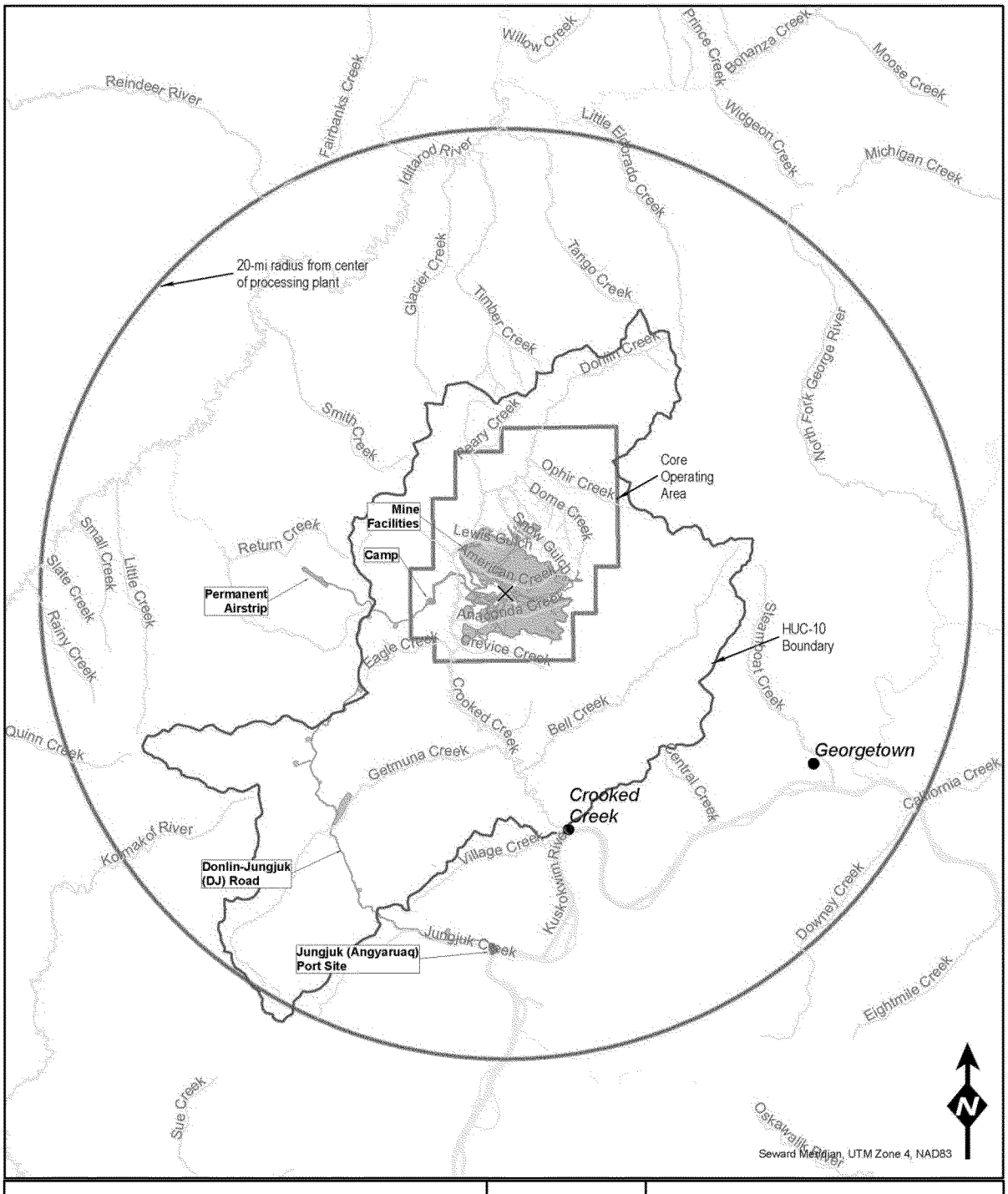
This Human Health Risk Assessment (HHRA) evaluated the likelihood of adverse effects to subsistence populations that could be exposed to constituents released into the environment by the operation of the proposed Donlin Gold, LLC (Donlin) project in remote western Alaska (Project). The proposed Project would build mining and milling facilities at the mine site and other associated transportation and fuel facilities.

The purpose of this HHRA was to address a concern expressed in comments on the *Donlin Gold Project: Draft Environmental Impact Statement* (DEIS; United States Army Corps of Engineers [USACE] 2015), which is to evaluate potential addition of metal constituents from the proposed Project to the environment in areas where subsistence populations live and harvest wild foods near the proposed Core Operating Area of the Project (Figure 1.1-1). A health-protective approach was taken to address this concern, in that conservative exposure assumptions were used to calculate potential incremental risk to subsistence populations that may be hunting, gathering, or living in the area around the Core Operating Area. A deterministic-based computation of non-cancer and cancer risks was completed following Alaska Department of Environmental Conservation (ADEC) and United States Environmental Protection Agency (USEPA) guidelines for HHRAs.

Risk assessments are not intended to predict the actual risk for an individual person. Rather, they provide conservative estimates of risk with an adequate margin of safety, according to USEPA guidelines for the protection of the majority of people that may potentially come into contact with constituents. General USEPA guidelines include 1989, 2000, and 2009a; other specific guidance incorporated is referenced within the HHRA.

Incremental risk from the proposed Project was raised as a concern in the DEIS comments. Both background (i.e., baseline) sources and Project-related air emissions would contribute to future environmental exposure. Therefore, the HHRA includes both a baseline estimate of potential risk and an incremental risk estimate from future Project-related sources.

This HHRA follows the format outlined in the ADEC Risk Assessment Procedures Manual (ADEC 2015), which describes five components of an HHRA: data evaluation, exposure assessment, toxicity assessment, risk characterization, and uncertainty assessment.



2. SELECTION OF CONSTITUENTS OF POTENTIAL CONCERN

Potential exposure to mercury (associated with mining activities) is a health concern expressed by stakeholders. Natural and anthropogenic sources of mercury currently exist in the environment in and around the proposed Core Operating Area. Proposed mining and processing activities could increase mercury levels in upland and aquatic habitats surrounding the Core Operating Area through point source (e.g., stack) and fugitive dust emissions. These concerns were raised in comments on the DEIS. A focused "Mercury Workshop" was held in December 2016 to address concerns and comments on Project-related mercury impacts. As a result of the December 2016 Mercury Workshop, Donlin Gold proposed to complete a focused, health-protective HHRA to address comments regarding this constituent.

The geochemistry of baseline soils and potential dust sources was evaluated in the DEIS (see Section 3.2.3.2.4 of USACE 2015). The analysis estimated potential dust deposition from the mine site on future soil levels. This analysis suggested that other metals of potential concern for soil quality include antimony and arsenic (Section 3.2.3.2.4 of USACE 2015). Therefore, antimony and arsenic were identified as constituents of potential concern (COPCs) and were evaluated in this HHRA.

3. CONCEPTUAL SITE MODEL

As part of the HHRA, a conceptual site model (CSM) was developed to describe the relationship between the sources of COPCs and potential exposure to human populations. The CSM is a representation of the relationship between COPC sources and exposed populations. It characterizes the distribution of COPCs across the study area and identifies all potential exposure pathways and migration routes. Figure 3.1-1 illustrates the sources, pathways, exposure routes, and exposed populations that the HHRA evaluated quantitatively. Each of these components is discussed in detail below. In addition, a description of the spatial and temporal boundaries of the HHRA is included, as these aspects also define the scope of the HHRA.

Complete and incomplete pathways of exposure were also identified in the CSM. Complete and potentially significant exposure pathways were identified and were evaluated quantitatively in the HHRA, while incomplete and/or non-significant pathways were eliminated from further consideration.

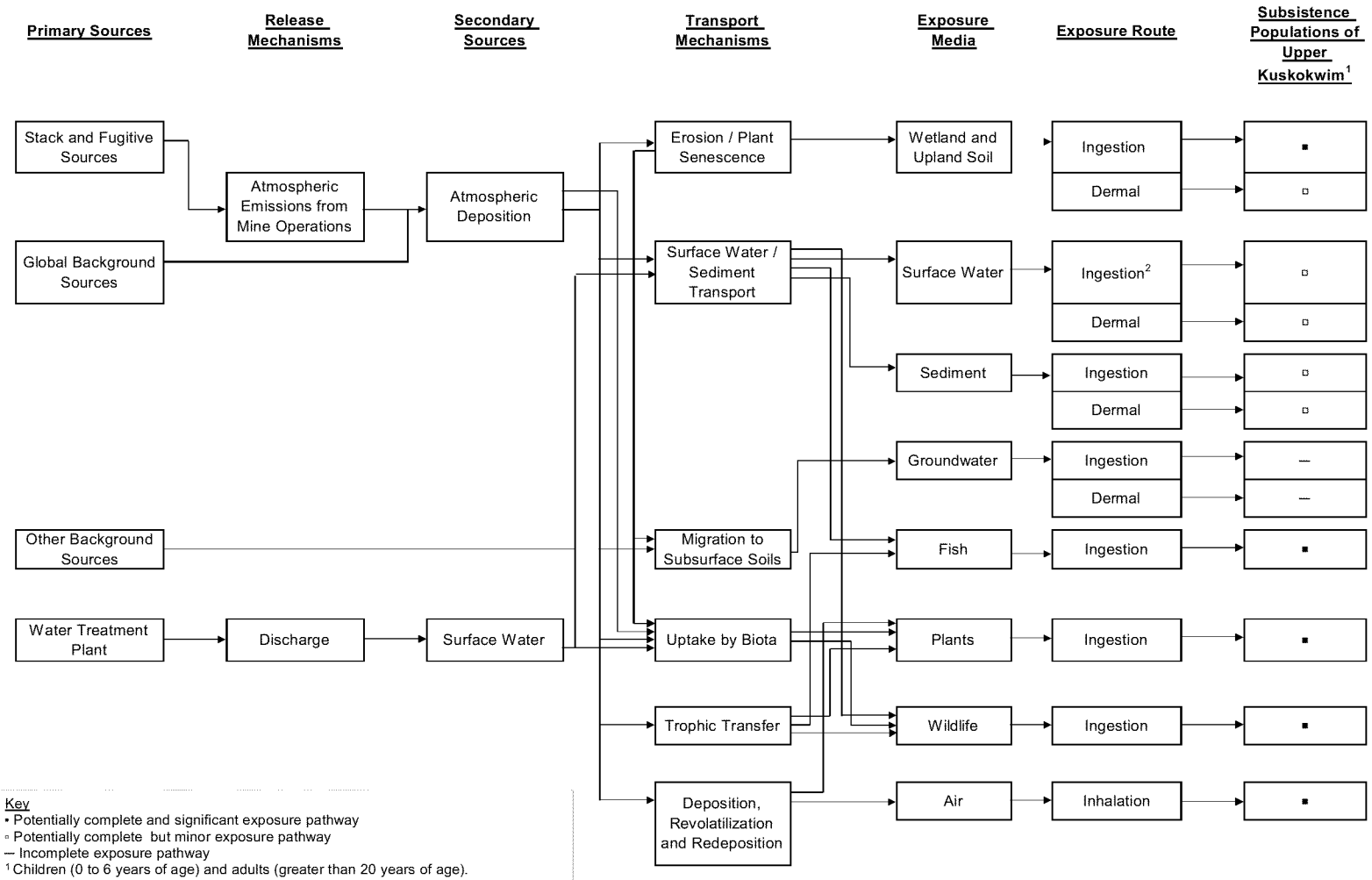
3.1. Temporal Boundary

The temporal boundary selected for the HHRA was determined by the periods during which planned Project activities would occur and have potential to affect the health of human receptors. The HHRA calculated risks for the last year of the mine life (year 27), when the increase in COPC concentrations in the environment from the proposed Project would be expected to be highest. The planned 27-year-based exposure represents the temporal boundary for maximum estimated releases from the proposed Project.

The potential exists that receptors may continue to be exposed to constituents after Project-related releases cease. This is due to the timeframe in which COPCs might move through the food web after the end of mine life and the longevity of fish and other species that people may consume. However, post-closure exposure is expected to be lower than during the operational mine life when proposed Project-related emissions will be highest, thus maximum potential exposure to proposed Project-related sources of COPCs were captured in the HHRA.

3.2. Spatial Boundary

The spatial boundary selected for the HHRA was determined by the proposed Project's potential impact on the health of human receptors. The HHRA considers exposures within a 20-mile radius outside the Core Operating Area, which represents the spatial extent of the mercury air deposition model (Environ 2015). This spatial area encompasses the watersheds in the vicinity of the proposed Project where the majority of deposition is estimated to occur from the proposed Project.



Human Health Conceptual Site Model

Donlin Gold Project

FIGURE **3.1-1**

This spatial boundary, referred to henceforth as the "study area," represents the highest exposure area for subsistence populations that could harvest wild foods in this area. It is within this boundary that risks associated with the individual exposure activities (e.g., hunting and gathering) were quantified in the HHRA. Even though subsistence gathering and hunting activities also occur outside this boundary (Brown et al. 2012), exposure to food items was conservatively assumed to occur entirely within this spatial boundary to estimate the maximum incremental exposure due to emissions from the proposed Project.

3.3. Primary Sources and Transport Pathways

The primary sources of COPCs include fugitive and point source emissions generated by the proposed Project, and for mercury, existing deposition from global background sources. Other sources of COPCs that may be generated by the proposed Project are not expected to be transported or deposited outside the Core Operating Area. Surface water that comes in contact with mining infrastructure ("contact water") will be treated prior to discharge to levels at or below USEPA and ADEC surface water criteria. Groundwater that could be impacted by the proposed Project is not a source of drinking water in the surrounding environment. As described in Section 3.7.3.2.3 of the DEIS (USACE 2015), groundwater impacts are anticipated to be limited to the American Creek drainage in the footprint of the Waste Rock Facility and extending to the pit, in the immediate vicinity of the pit, and groundwater in the Anaconda Creek drainage below the tailings storage facility that is captured in the seepage recovery system. As future development of groundwater wells is not reasonably anticipated in this area, consumption of groundwater is not a complete human health exposure pathway. Therefore, groundwater was not included as a source for the HHRA.

Atmospheric emissions from the proposed Project have the potential to enter the atmosphere, travel some distance, and be inhaled by receptors or settle as dustfall where they can reside in different media such as soil, vegetation, and subsistence foods. Modeled mercury deposition from the processing plant point and fugitive sources was estimated for watersheds surrounding the Project (Environ 2015). The model showed that mercury would be emitted predominantly as gaseous, elemental mercury [Hg(0)], and the mercury that is deposited would consist of Hg(0), oxidized mercury [Hg(2+)], and particulate mercury [Hg(p)], with the majority in the particulate form. Oxidized mercury (Hg(2+) is more likely to be methylated upon deposition and is a minor component (approximately 1 to 2 percent) of the total deposited.

Antimony and arsenic will also be emitted into the atmosphere from proposed Project point and fugitive sources. Transport and deposition mechanisms of these constituents are expected to be comparable to elemental mercury transport, as summarized in Table 3-9 from Environ (2015).

The particulate-bound mercury emissions estimate used by Environ (2015) was derived from the fugitive dust inventory. Arsenic and antimony emissions were also calculated in the fugitive dust emissions inventory, as presented in Air Sciences (2016), and scaling factors were derived from the relative mass emissions to estimate arsenic and antimony deposition. The mass loading values for arsenic and antimony compared to mercury produce scaling factors of 10.1 for antimony and 320 for arsenic. Background concentrations of arsenic and antimony bound to particulate matter less than 2.5 microns ($PM_{2.5}$) are assumed to be negligible.

3.4. Exposure Pathways

Human exposure pathways are the routes by which people are exposed to COPCs. COPCs must be absorbed into the body of humans, animals, or plants to cause a toxic response. Absorption can occur through oral ingestion, dermal absorption, or inhalation. The exposure pathways that may exist between COPCs and human receptors depend on many factors which may be direct, indirect, or both.

An exposure pathway must include a source (e.g., proposed Project-related atmospheric emissions), a mechanism of release and transport pathway to an affected medium (e.g., atmospheric deposition onto soil), a receptor (e.g., subsistence harvester), and an exposure route (e.g., incidental soil ingestion during harvesting). An exposure pathway is deemed complete when all components are present. If one or more components are missing, then the pathway is incomplete. Complete pathways may be deemed insignificant as the level of exposure may be too low to be quantifiable or a health concern.

Exposure pathways selected for quantitative evaluation in the HHRA include:

- Inhalation of COPCs present in air;
- Incidental ingestion of COPCs bound to soil; and
- Ingestion of COPCs associated with biota (fish, plants, and wildlife) that may be harvested in the study area.

Dermal absorption of COPCs was considered a potentially complete but negligible pathway and was not quantified in the HHRA. Absorption of antimony, arsenic, and some forms of mercury (e.g., Hg[0]) following dermal exposure is negligible in humans and animals. Robust exposure data to quantify dermal absorption of these COPCs are lacking. The predominant form of COPCs present on the surfaces of biota or associated with primary media (soil, sediment) would be inorganic forms (e.g., Hg(0)). Further, the predominant exposure pathway of organic forms (e.g., MeHg) is through oral ingestion in the diet (USEPA 2009a).

3.5. Exposure Media

Exposure media include air, soil, and biota that could be collected and ingested by subsistence populations. Because not all individual species or wildlife trophic components of an ecological system are practical or potentially necessary to quantify incremental risk, representative species were chosen. From a technical risk perspective, the criteria used to select representative subsistence food exposures included consideration of:

- Potential presence and abundance in the study area (i.e., plant species abundant in the study area, and fish/wildlife that have limited migration patterns and small home ranges were preferred over species that are migratory or that move in and out of the study area);
- Physiological characteristics of wildlife species that would result in relatively higher rates of exposure or bioaccumulation of COPCs (e.g., dietary sources that would occur within the study area, and/or higher trophic levels that would accumulate mercury to a higher degree than other species); and
- Subsistence consumption patterns (species that are harvested/consumed more frequently were preferred over species with low consumption rates).

Based on review of the subsistence food items harvested by the Upper Kuskokwim group (see Section 3.21 in the DEIS), and accounting for physiological considerations and residency in the study area, the HHRA quantified the incremental risk of subsistence population consumption of northern pike (representing residential fish species consumption), beaver (representing small mammal consumption), mallard ducks (representing wild bird consumption), and blueberries and cranberries (representing wild berry consumption).

3.6. Exposure Populations

The HHRA focused on receptor populations that would have the highest potential to be exposed. The receptor population with the highest potential to be exposed in the study area included subsistence receptors. Subsistence receptors are individuals who consume locally caught and harvested food items (e.g., locally caught fish) as major sources of food. Thus, their intake rates of these food items are typically higher than the general population (USEPA 2011). These populations are also exposed via direct contact with abiotic media (e.g., incidental ingestion of soil and inhalation of particulate material).

Subsistence groups live in and around the study area, and harvest, hunt, and fish throughout the region. Subsistence harvest patterns described in the DEIS focus on community profiles from nine subregions. Four of those subregions harvest food near and within the study area: Upper, Central, Lower-Middle, and Lower Kuskokwim groups. Only the Crooked Creek village and Georgetown subsistence populations live in the study area. The Crooked Creek village is located south-southeast of the proposed Project, and Georgetown is to the east-southeast of the proposed Project. The

predominant wind direction in the study area, when blowing to the south, is to the south-southeast.

The subregions share a common ecology and some common harvest patterns. Harvesting, hunting, and fishing ranges of subregion groups are large and only include a portion of the study area. Exact harvests from one area compared to the next were not reported, and areas of harvest also vary seasonally and annually. For the purposes of the HHRA, a generalized subsistence group was developed to quantify risks. This generalized subsistence group was assumed to live in the study area (therefore evaluating year-round inhalation and soil ingestion risks), and to hunt, fish, and gather only within the study area for beaver, mallard ducks, berries, and northern pike. Inhalation risk was specifically quantified for air concentrations predicted for the Crooked Creek watershed, which includes the Crooked Creek village area. Quantities of wild foods harvested and consumed were based on data presented for the Upper Kuskokwim subregion group, as this group lives nearest and most frequently harvests nearest to the proposed Project. Because these groups also hunt, fish, and gather outside the study area to varying extents, these assumptions overestimate the exposure frequency and duration in the study area. This is a conservative analytical (i.e., health protective) approach that ensures that potential effects on human health due to exposures via inhalation, as well as consumption of wild food collected within the study area, are not underestimated.

The HHRA evaluated children and adults in the subsistence population: children between 0 and 6 years old, and adults greater than 20 years of age. Children are often most susceptible to constituents due to their small body size relative to their ingestion rates compared to other human life stages.

4. DATA EVALUATION

4.1. Data Sources and Usability

Selection of data for this HHRA followed as closely as possible the ADEC guidelines for data usability. Both Project-specific studies, collected for the purposes of baseline environmental characterization, and other regional datasets, were used to develop baseline exposure point concentrations (EPCs) for various media, as well as used to develop bioaccumulation factors (BAFs) and predict future methylation rates.

Sources of data used in this HHRA are summarized in Appendix A. Only data that gave accurate chemical-specific concentrations were used. The datasets collected for the proposed Project included method detection limits and sample quantitation limits below ADEC and/or USEPA criteria for their respective media, as applicable and included explanations for qualified data. For data reported by other sources, method and quantitation limits were not always reported. However, only sources of information that were peer reviewed and/or that followed USEPA guidelines for HHRAs, which include a data adequacy evaluation, were used in this HHRA.

In addition to Project-specific sources of information, data collected regionally, near to the study area, were considered. Regional datasets included:

1) Bureau of Land Management (BLM) study on mercury, arsenic, and antimony in aquatic biota from the Middle Kuskokwim River Region, Alaska (Matz 2012).
2) United States Geological Survey (USGS) studies of metals in sediment, stream water, and fish collected within and surrounding the Middle Kuskokwim River in southwestern Alaska (Wang 1999; Bailey and Gray 1997; Bailey et al. 2002; and Gray et al. 1996, 1997, 2000).
3) BLM's Final Remedial Investigation Report (RI Report), authored by Ecology and Environment, Inc. (E&E 2014). This was specifically prepared for the Red Devil Mine site.

Of the regional reports available, only the RI Report disclosed raw dataset(s), though reporting limits and method detection limits were not reported. The RI Report was subsequently used for comparisons of BAFs. Donlin requested that the BLM provide mercury, arsenic, and antimony analytical data and sample collection details (raw data) for sample locations referenced in a report prepared for the BLM by the US Fish and Wildlife Service titled "*Mercury, Arsenic, and Antimony in Aquatic Biota from the Middle Kuskokwim River Region, Alaska, 2010-2011, Interim Report*" (Matz 2012). The BLM responded that they anticipate the raw data collected for the study will be available to the public in the future but only after the BLM has decided on a final remedial action for the Red Devil Mine. The BLM consequently did not anticipate the data being available until sometime after 2017, and the data were not available for use in the HHRA.

The Alaska Department of Health and Social Services (ADHSS) conducted MeHg testing of hair samples from pregnant women in selected communities, including the Upper Kuskokwim River region communities, between May and September 2012. A total of 186 hair samples were obtained from eight Upper Kuskokwim River communities. As reported in the DEIS (Section 3.22.2), the median hair mercury level for the study population was 0.510 parts per million (ppm), with a range of 0.030-3.707 ppm. Every study participant had a hair mercury level that was below the ATSDR No-observed-adverse-effect-level (NOAEL) (15.3 ppm), as well as the Environmental Public Health Program cut-off for follow-up (5 ppm) (ADHSS 2013, as cited in the DEIS). Overall, increasing median hair mercury levels were weakly associated with increasing age (r^2=0.28, $p<0.05$). Mercury hair levels did not differ significantly between communities.

Hair sampling as a method of biomonitoring offers a less invasive method to monitor for possible exposure to mercury and other bioaccumulative substances. However, as noted by ATSDR (2003), sampling and analysis of hair samples has many limitations, notably, standard procedures have not been published for collecting, washing, and analyzing hair samples. There is no developed method to relate dose (as is quantified in HHRAs) to concentrations in hair, in part because hair analytical results cannot pinpoint the sources of constituents (e.g., ingested versus dust that falls onto hair) that were detected. Other assumptions in the HHRA, such as exposure frequency and duration (selected to generally overestimate potential exposure), may not match actual frequency and duration of the populations measured for mercury in hair. For these reasons, hair data collected by Newfields (2015) was not used to quantify baseline risks in the HHRA. However, overall outcomes of the hair sampling study are compared to risk results (Section 7) to provide additional context in evaluation of baseline exposure and risk.

5. EXPOSURE ASSESSMENT

Exposure assessment is the process of determining magnitude, frequency, duration, and route of exposure to COPCs. The results of the exposure assessment include estimates of chemical intake that, combined with chemical-specific toxicity information, characterize potential risks. The calculation of chemical intake, or dose, requires estimation of EPCs, and estimation of intake rates. Because future (and some baseline) EPCs need to be estimated, future potential for mercury methylation was evaluated (Section 5.1), and BAFs were developed (Section 5.2), to understand potential future mercury, antimony, and arsenic bioavailability and uptake into biota.

5.1. Current and Future Methylation

Mercury is a naturally occurring, reactive metal that exists in various chemical forms under different environmental conditions. Mercury is capable of existing in three oxidation states: 0, +1, and +2. In the atmosphere, mercury exists predominantly in the elemental (Hg[0]) form, while most mercury in water, soil, sediments and biota is in the form of inorganic and organic mercury complexes. Inorganic mercury is regionally enriched as evidenced by cinnabar ore deposits of sufficient grade to support development of the nearby Red Devil Mine.

Anthropogenic emissions, particularly atmospheric emissions, have the potential to increase mercury loadings to the environment. Following atmospheric deposition, various complex and interrelated environmental factors in terrestrial and aquatic ecosystems determine mercury fate and transport, including the potential for mercury methylation. MeHg is of particular concern because: (1) it is the most toxic form of mercury and (2) it bioaccumulates in food chains to a greater extent than inorganic forms of mercury. Formation of MeHg is mediated by certain naturally occurring bacteria, and rates of methylation are dependent on a variety of factors affecting the metabolism of these organisms. Current concentrations of total mercury (THg) and MeHg were measured in the study area (ARCADIS 2014). The ratio of THg to MeHg, referred to as the baseline methylation rate, was computed. Future methylation potential must be estimated to understand future THg and MeHg concentrations and bioaccumulation potential into biota. This section discusses the following key topics:

- Review of factors affecting methylation rates;
- Review of baseline methylation potential;
- Literature review/discussion of approaches to estimate future methylation, including consideration of mercury speciation models suggested by USEPA (Spreadsheet-based Ecological Risk Assessment for the Fate of Mercury [SERAFM] and Dynamic Mercury Cycling Model [D-MCM]);

- Analysis of sulfate loading, different sources and geochemical forms of mercury, and other environmental factors as they may affect future methylation rates;
- Basis and rationale for the approach to predict future mercury methylation potential; and
- Estimates of future THg and MeHg concentrations in soil and surface water based on the analyses described.

5.1.1. Environmental Factors Affecting Methylation Rates

The rate of conversion of THg to MeHg is the methylation rate. Commonly, this rate is represented by the ratio of MeHg to THg (or %MeHg). The methylation rate (%MeHg) is dependent on complex environmental factors (USEPA 1997b, Environment and Climate Change Canada 2016, Frohne et al. 2012, Houben et al. 2016, Scudder et al. 2009, Ullrich et al. 2001). Numerous studies have attempted to quantify driving factors, sometimes with conflicting results and rarely with quantifiable relationships. Subsequent sections draw from comprehensive reviews of mercury science in the cited literature.

In the aquatic and terrestrial environments, sulfate reduction by sulfate reducing bacteria (SRB) is the primary mechanism for generating MeHg from divalent mercury (Hg[2+]). Consequently, inhibiting or promoting the activity of the SRB will have a strong effect on methylation rates. Metabolism by SRB is the primary mechanism for the conversion of mercury into MeHg; therefore, the primary controls on methylation are the presence/absence of an appropriate mercury pool and the presence/absence of an oxygenated environment. Once these two conditions are met, the secondary controls are the availability of nutrients and metabolic materials, including sulfate and organic carbon. Additional controls are related to factors controlling the rate of microbial metabolism, such as temperature and pH. Finally, the rates of demethylation reactions (e.g., photodemethylation) can be locally important, but are likely to be tertiary order controls.

5.1.1.1. Primary Mechanism: Mercury Content

The availability of Hg(2+) is a critical limiting factor for the production of MeHg; however, direct measurement and reporting of Hg(2+) concentrations are rare. Concentrations of MeHg and THg are typically positively correlated, but the relationship is not universal, as shown by Watras et al. (1995) and Gray et al. (2002). Little research has focused on measuring methylation rate changes at very low or very high concentrations of THg, thus testing where the concentration of mercury would be a limiting factor in methylation. Consequently it is not known if there is a lower or upper bound on mercury concentration that inhibits methylation.

5.1.1.2. Primary Mechanism: Oxygen Content (Redox Controls)

Sulfate- and iron-reducing microbes are most efficient in anoxic environments, but can respire in sub-oxic environments (Environment and Climate Change Canada 2016, Baumgartner et al. 2006). Aquatic environments with oxidation-reduction (redox) values of -0.05 to -0.22 Eh (V) are typically required for biotic methylation suggesting that saturated wetlands, lake sediments, and hypolimnetic lakes are prime locations for methylation (Environment and Climate Change Canada 2016, Eckley and Hintelmann 2006). Obrist (2012) found that in a forested system, THg decreased with soil depth (to 20 centimeters [cm]), but MeHg stayed approximately the same, resulting in a weak positive correlation (r2 of <0.3, slope of 0.0006) in MeHg to THg content with soil depth (Table 5.1-1), possibly reflecting declining oxygen content. In aquatic environments, methylation is less prevalent in environments with higher flow and low hydraulic retention (St. Louis et al. 1994). In general, in-river methylation is typically a negligible component of the MeHg budget for creeks (St. Louis et al. 1994, Berndt and Bavin 2012).

Upland soils tend to be more oxygenated than lowland/wetland soils and consequently, upland soils have a lower MeHg generation potential. In general, wetlands are considered to be net sources of MeHg, and uplands are sinks for MeHg. St. Louis et al. (1996) concludes that boreal forest catchments can be sinks for THg regardless of the proportion of wetlands in the catchment. Contradicting St. Louis et al. (1996), Brigham et al. (2009) studied streams in Oregon, Wisconsin, and Florida, and identified that there was a strong positive correlation between the proportion of wetlands in a basin and the filtered THg and MeHg concentrations in surface water. The contrasting studies illustrate the complex and site-specific nature of mercury fate and transport and the lack of consistent, predictable mechanisms to predict mercury flux.

TABLE 5.1-1: OBRIST (2012) SUMMARY: THG, MEHG, AND MEHG/THG RATIOS IN SOIL

Mercury Species	Litter Oi	Litter Oe	Litter Oa	Soil 1-20 cm	Soil >20 cm
THg (µg/kg)	34	81	190	43	33
MeHg (µg/kg)	0.12	0.25	0.23	0.07	0.04
MeHg/THg (%)	0.29	0.33	0.12	0.16	0.11

Source: Obrist (2012).

5.1.1.3. Secondary Mechanism: Sulfate

Sulfate concentrations and methylation rates can be both positively and negatively correlated. Low concentrations of sulfate can limit reduction reactions by SRB, while high concentrations of sulfate can slow down or inhibit the reactions due to the production of sulfide metabolic waste products, mercury-sulfur (Hg-S) complexes, which compete for available Hg(2+) (Environment and Climate Change Canada 2016).

While a lower limit of sulfate concentration has not been explicitly identified in research, it has been noted that the optimal sulfate concentration for Hg(2+) methylation to occur ranges from 1 to 29 milligrams per liter (mg/L) (Environment and Climate Change Canada 2016, Drott et al. 2008, Benoit et al. 1999, Benoit et al. 2001). Up to about 50 mg/L sulfate can stimulate SRB (Houben et al. 2016), indicating that until that concentration is reached; sulfate is a rate-limiting factor. One study noted that the concentration of MeHg increased in proportion to the concentration of free sulfide (S2-) ions, reaching a maximum at 1.8 milligrams S2- per gram of sediment. Above this maximum, the concentration of MeHg decreased, perhaps due to formation of dimethylmercury (Craig and Moreton 1986). An additional complication is that circum-neutral pH Hg-S complexes are highly bioavailable to mercury-methylating microbes.

Generally, sulfate concentrations below 50 mg/L will limit the metabolic rate of SRB. At higher sulfate concentrations, the formation of sulfide phases may also limit the metabolic rate of SRB. Increases to sulfate concentration in natural systems with low concentrations of sulfate will result in increases in the concentration of MeHg. The reviewed research indicates that the factor increase in MeHg is 0.5 to 1.0 times the factor increase in sulfate, as described in the subsequent paragraphs.

Studies involving enhancing sulfate concentrations in a wetland/peatland (Jeremiason et al. 2006 and references contained therein) demonstrated that the MeHg response to enhanced sulfate concentrations was approximately proportional; i.e., a 10-fold increase in porewater sulfate concentration was accompanied by a 5- to 7-fold increase in porewater MeHg concentrations. In these studies, the background sulfate concentrations were initially low (at or below 0.05 mg/L) until the initial spike. Subsequent spikes in the summer did not increase the sulfate or MeHg concentrations in porewater due to warmer temperatures enhancing sulfate reduction rates.

Building on previous work performed at the Marcell Experimental Forest in northern Minnesota, a study by Coleman Wasik et al. (2012) investigated the effect that enhanced sulfate concentrations had on MeHg concentrations in peat/wetland porewater. Artificially increasing the sulfate concentration from approximately 2 to 2.9 mg/L in 2006 and from less than 0.25 to 3.8 mg/L in 2008 resulted in an increase of MeHg concentrations from approximately 2 nanograms per liter (ng/L) to 4.3 ng/L in 2006 and to 3.6 ng/L in 2008. The proportion of THg attributed to MeHg also increased after sulfate addition. One of the study's conclusions was that increasing the sulfate concentration in porewater by four times led to an MeHg increase of a similar magnitude in both the peat porewater and the peat solid.

Data presented in Mitchell et al. (2007) indicate that this consistent positive relationship between sulfate concentration and MeHg concentration is not observed at higher sulfate concentrations. In their study, high %MeHg results occurred at sulfate concentrations below 5 mg/L, while above 7 mg/L, the %MeHg decreased to near zero

at 47 mg/L sulfate, supporting the assertion that high concentrations of sulfate can slow down or inhibit the reactions due to the production of sulfide metabolic waste products, Hg-S complexes, which compete for available Hg(2+) (Environment and Climate Change Canada 2016).

Overall, the literature review indicates that increases in sulfate concentrations are commonly accompanied by increases in MeHg concentrations. Although variable, literature suggests that in systems where sulfate is initially at low concentrations, methylation rates may increase up to a similar magnitude as the increase in sulfate concentrations.

5.1.1.4. Secondary Mechanism: Organic Carbon Availability

Organic carbon concentrations and methylation rates have displayed both positive and negative correlations. Organic carbon is a nutrient for SRB microbes; therefore, higher concentrations are often accompanied by increased methylation rates. However, dissolved organic matter (DOM) forms complexes with Hg(2+). Smaller organic ligands can easily pass through microbe cell walls and are used as electron donors enhancing reaction rates, while higher molecular weight DOM cannot pass through the cell wall inhibiting the reaction rate (Environment and Climate Change Canada 2016, Golding et al. 2002, Barkey et al. 1997). At moderate to high DOM concentrations (9-30 mg/L; Miskimmin et al. 1992), the influence of other parameters such as pH may be more important, making correlations difficult to determine.

Generally, at lower concentrations of DOM, an increase in DOM is accompanied by an increase in the rate of methylation. However, high concentrations of DOM can decrease methylation rates (Ullrich et al. 2001).

5.1.1.5. Other Secondary Mechanisms: pH, Temperature

Temperature affects microbial and abiotic reactions. The metabolism of most microbes is positively correlated up to about 35 degrees Celsius (Winfrey and Rudd 1990). Consequently, the methylation rates in aquatic ecosystems are positively correlated with the temperature of the aquatic environment, with the highest methylation rates occurring in mid- to late summer (Ullrich et al. 2001, Environment and Climate Change Canada 2016, Bodaly et al. 1993). Methylation can occur year-round if the sediment does not freeze, resulting in a winter contribution to many frozen lakes. However, streams in the study area freeze, beginning in about early-October.

Results from studies carried out in surface waters indicate that there is a negative relationship between pH and methylation rate; as pH decreases, the proportion of THg that is MeHg increases. Most natural ecosystems do not have pH values low enough to severely inhibit SRB metabolism. A commonly postulated reason for increased methylation rates at lower pH is decreased binding of Hg(2+) by dissolved organic carbon (DOC) resulting in greater concentrations of Hg(2+) being available for

methylation reactions (Environment and Climate Change Canada 2016, Kelly et al. 2003). However, Ullrich et al. (2001) noted that the relationship is complicated as pH can affect multiple biogeochemical systems.

Multiple studies have attempted to quantify the relationship between pH and mercury. While some studies found that pH was the dominant factor affecting the methylation rate (Miskimmin et al. 1992), others found only a weak correlation (Frohne et al. 2012).

As pH drops from 7 to 5, the methylation rate typically increases (Miskimmin et al. 1992, Ullrich et al. 2001). It is not clear if this increase is a function of increasing methylation rates or decreasing demethylation rates, or some combination of the two. pH values below 5 were commonly associated with decreasing %MeHg (Mitchell et al. 2007). As baseline pH values reflected in proposed Project baseline data for terrestrial ecosystems are below pH 5, the methylation rate may be limited slightly.

5.1.1.6. Tertiary Mechanism: Photodemethylation Reactions

Photodemethylation is an abiotic process prevalent in the surface waters of lakes. Demethylation can also occur when mercury-resistant microbes reduce the MeHg to Hg(0). Oxidative demethylation can also occur and is thought to be mediated by anaerobic microbes. The largest of these three methods is photodemethylation (Environment and Climate Change Canada 2016, Amyot et al. 2001, Garcia et al. 2005).

Visible and ultra-violet wavelengths can produce both Hg(0) and Hg(2+) from MeHg. Hg(0) can easily escape the aquatic environment to become atmospheric Hg(0) while Hg(2+) has a longer residence time and has the potential to be re-methylated (Environment and Climate Change Canada 2016). However, in most instances the rate of demethylation is lower than the rate of methylation in lakes.

5.1.2. Baseline Methylation Rates – Soils

While there is a high degree of variability in the methylation rates reported for different wetlands studied, the average values in wetland porewater for THg, MeHg, and %MeHg are similar between geographically diverse studies in northern North America. In Canadian Arctic wetlands, soils typically have low THg concentrations, ranging between 10 and 250 micrograms per kilogram (µg/kg). MeHg concentrations are also low, ranging between 0.34 and 3.07 µg/kg, with %MeHg less than 1% (AANDC 2012). Other studies have identified porewater MeHg concentrations between <0.6 and 0.63 ng/L and %MeHg between <7 and 13.3% as baseline conditions for peatlands and wetlands (Coleman Wasik et al. 2012, St. Louis et al. 1994). In a more intensive sampling program to identify peatland hotspots of high %MeHg in the interior, Mitchell et al. (2007) sampled four wetlands and reported the following median porewater ranges: THg 5.5 - 13.3 ng/L, MeHg 0.35 - 0.62 ng/L, and %MeHg 2.7 - 15.9%.

Summary statistics for upland and wetland (or lowland) soil samples collected in the HHRA study area are presented in Tables 5.1-2 and 5.1-3. In general, the wetland sites had somewhat higher MeHg and %MeHg than the upland samples. However, median values for upland and wetland datasets were similar and varied by less than one standard deviation. A similar observation was made by ARCADIS (2014) in evaluating paired upland and wetland samples collected in the study area.

TABLE 5.1-2. SUMMARY STATISTICS FOR UPLAND SOILS COLLECTED IN THE HHRA STUDY AREA

	Min	Median	Max	Count >DL	SD	DL
THg (µg/kg)	26.5	94.9	1740.0	45	270.4	0.2
MeHg (µg/kg)	0.10	0.29	3.95	10	1.21	0.01
MeHg/THg (%)	0.15	0.47	2.59	10	0.77	NA
TOC (%)	4.6	18.0	70.0	10	20.2	NA
pH	4.00	4.55	6.70	10	0.75	NA
C:N ratio calculated	14.9	40.7	77	10	21.3	NA
Sulfate (mg/kg)	10.0	11.0	12.0	2	NA	10
Sulfide (mg/kg)	NA	5.32	NA	1	NA	5
%MeHg/ SO_4	0.00015	0.00047	0.00259	10	0.00073	NA
THg/ SO_4	0.00265	0.00797	0.17400	10	0.0497	NA
MeHg/ SO_4	0.000010	0.000029	0.000395	10	0.00012	NA

TABLE 5.1-3. SUMMARY STATISTICS FOR WETLAND SOILS COLLECTED IN THE HHRA STUDY AREA

Parameter	Min	Median	Max	Count >DL	SD	DL
THg (µg/kg)	49.5	106.5	1850.0	10	528.5	0.2
MeHg (µg/kg)	0.17	0.60	3.98	10	1.09	0.01
MeHg/THg (%)	0.09	0.54	1.83	10	0.47	--
TOC (%)	6.6	28.0	55.0	10	15.6	--
pH	4.30	4.50	4.80	10	0.17	--
C:N ratio calculated	18.5	33.6	57.6	10	10	--
Sulfate (mg/kg)	12.0	14.5	17.0	2	NA	10
Sulfide (mg/kg)	NA	NA	NA	0	NA	5
%MeHg/ SO_4	0.00009	0.00047	0.00183	10	0.00047	--
THg/ SO_4	0.00438	0.01065	0.185	10	0.0524	--
MeHg/ SO_4	0.000017	0.000060	0.000234	10	0.000068	--

NA = *not applicable because the value cannot be calculated.*

-- Detected in all samples.

Study area data were compared to other data collected in the region and to a study (Obrist 2012) that evaluated mercury ratios across 14 US forests, described below.

Median THg was lower and %MeHg was higher in study area soils compared to background samples at the Red Devil Mine (Bailey et al. 2002) and in material from the Sleetmute Quadrangle, Alaska (Miller et al. 1998). Rock data from the Sleetmute Quadrangle represent primarily upland outcrops, while the stream sediment data represent lowland sediments (Miller et al. 1998). The rock samples had a mean THg content of 191 ppm (191,000 µg/kg), while stream sediments from the same region had a mean THg content of 0.14 ppm (141 µg/kg). The slightly higher values in the Sleetmute Quadrangle (relative to the study area) may reflect naturally elevated mercury concentrations in a localized area around the Red Devil Mine mercury deposit. The mean concentrations of THg and MeHg in background soil samples near the Red Devil Mine were 400 and 0.8 µg/kg, respectively (Bailey et al. 2002), and MeHg/THg ratios were between approximately <0.01 and 1.0% (median ratio approximately 0.2%).

Median THg and %MeHg values for study area upland and wetland soils were higher than MeHg data presented from 12 forest sites by Obrist (2012). One conclusion of this study was that MeHg concentrations varied through the leaf litter and into the soil profile. Typically, the more decomposed leaf litter (Oa layer) had the highest MeHg concentrations, the highest MeHg/carbon ratios, and the highest MeHg/THg ratios (Table 5.1-1). The study compared the 12 upland forest sites to other upland studies and found that MeHg in leaf litter was comparable, but MeHg in soils was on the low end compared to other studies. The relationship between THg and MeHg had a weak positive correlation (r^2 of <0.3) with a slope of approximately 0.0006 (MeHg/THg).

In summary, the Project baseline methylation rate of soils is slighter higher than results in other studies in the region, but the values are within the range of other background sites in North America. The median baseline methylation rate is estimated at 0.47% (Tables 5.1-2 and 5.1-3). ProUCL-generated means of THg and MeHg in soils (see Section 5.3.3) also suggest an overall, mean %MeHg of about 0.5%. However, a paired sample collection program (ARCADIS 2014) determined a mean %MeHg of closer to 1%.

5.1.3. Baseline Methylation Rates – Aquatic Ecosystems

Mercury can be dissolved or remain in a particulate state in aquatic systems. Aqueous inorganic mercury can consist of both free ions and inorganic mercury complexes. Aqueous organic mercury occurs either as covalently bonded organomercurials such as MeHg, or mercuric complexes with organic matter, such as humic substances.

The baseline water quality is very similar to unimpacted regional and continental values (Enos 2013, Weglinski 2016). Summary statistics are presented for surface water and sediment in Tables 5.1-4 through 5.1-7. Surface water and sediment data were divided into two datasets as described in the DEIS (USACE 2015): (1) Category I

(upgradient of mineralized areas) and (2) Categories II and III (within or downstream of mineralized areas) for purposes of discussion. Percent MeHg was similar between categories for both surface water and sediment datasets, as were minimum and median THg and MeHg. However, maximum THg and MeHg were higher in the Category II/III dataset. Also noted, sediment summary statistics were very similar to the soil data statistics (see Tables 5.1-2 and 5.1-3).

Methylation rates based on unfiltered baseline THg and MeHg surface water data were within regional ranges indicating no particular process that enriches or depletes mercury within the study area. The median %MeHg values are close to 2%; however, the ranges of %MeHg values were similar between soil and surface water datasets, implying similar processes and rates for both media. A linear relationship provides the best fit for baseline stream water samples (Figure 5.1-1), and the equation of the line is approximately:

$$MeHg = 0.0024*THg + 0.00000005;\ r^2 = 0.81,\ n = 28$$

The slope of the above equation implies that the overall %MeHg is approximately 0.24%. This does not agree with the average %MeHg for the water quality sample set (2.0%), because the majority of the data are clustered at low THg values and subsequently exhibit a much higher %MeHg. However, it does agree with %MeHg results from baseline soil samples, as well as sediment samples. Summary statistics for baseline sediment samples show median %MeHg, THg, and MeHg results very similar to the upland and wetland soil dataset, with a median %MeHg of 0.28.

The surface water sulfate values are also similar to the sulfate concentrations in soil samples, while sediment sulfate values were below the method detection limit.

Although a relationship has been observed between increased THg concentrations and MeHg concentrations in aquatic ecosystems, the relationship is not necessarily linear. For example, increasing the concentration of THg in laboratory mesocosms representing lakes resulted in increased MeHg concentrations (Orihel et al. 2007). Mesocosms were spiked with THg loads, resulting in concentrations between two and 12 times higher than background; however, while the measured response in sediment and water mercury concentrations mirrored the addition of mercury, the relationship was not directly linear.

Stream sediments are intermediate between uplands and wetlands, and encompass energetic, oxygenated environments and slower, organic-rich environments. Stream sediments represent the upstream inputs and outputs and can therefore vary along individual reaches.

Scudder et al. (2009) summarized THg and MeHg sediment results for hundreds of US streams in mined and unmined basins. Concentrations of THg in sediment ranged from 0.84 to 4,520 µg/kg. About 75% of THg results were less than 80 µg/kg. Concentrations

of MeHg ranged between 0.01 to 15.6 µg/kg. The %MeHg ratios varied between 0.02 to 41.0%, with unmined basins having higher %MeHg ratios than mined basins.

Stream water data summarized by Scudder et al. (2009) showed THg concentrations between 0.27 and 446 ng/L and MeHg concentrations of <0.010 to 4.11 ng/L. Some data were collected from mined basins, and hence the data represent a very broad set of unimpacted and impacted conditions. The %MeHg ratios varied between 0.02 and 81.5% with unmined basins having higher %MeHg ratios than mined basins.

Stream sediment samples collected from upstream and downstream of the Red Devil, Cinnabar Creek, and Red Top mercury mines in Alaska were found to have Hg(2+):THg ratios generally less than 5% and %MeHg less than 1% (Gray et al. 2000). The Red Devil Mine results showed that THg varied between approximately 200 and 3,000,000 µg/kg, while MeHg concentrations varied between approximately 300 and 4,000 µg/kg, indicating that the %MeHg decreased with increasing THg. In the unfiltered stream water samples, the MeHg concentrations were less than 1.5 ng/L and constituted less than 3% of the THg.

Regional stream sediment data from the Kuskokwim River, Holitna River, Red Devil Creek, and Crooked Creek were summarized by Wang (1999). THg concentrations were low in the Kuskokwim River and Holitna River samples with nine of the 10 sample sites having mean bulk THg values lower than 100 µg/kg and a mean MeHg concentration of 0.18 µg/kg. Red Devil Creek had bulk THg values from 500 to 150,000 µg/kg, and bulk MeHg concentrations of 1.75 µg/kg; the results were lower than those presented in Gray et al. (2000). Crooked Creek had a bulk THg value of 280 µg/kg and a bulk MeHg concentration of 3.12 µg/kg.

Water samples from the Kuskokwim River and Holitna River sites had mean THg and MeHg concentrations of 4.3 and <0.06 ng/L. The Red Devil Creek and Crooked Creek sites had THg concentrations of 236.5 and 10.5 ng/L and MeHg concentrations of 0.32 and 0.49 ng/L, five times lower than data presented in Gray et al. (2000).

Table 5.1-8 summarizes sediment and water mercury concentrations from three different scales of studies—continental, large river basin, and small river basin. In general, the broader continental data had lower mercury concentrations than the large river basin data, which had lower concentrations than the small river basin data. The sediment mercury results varied by a factor of 9 compared to the water mercury results which varied by a factor of about 5. However, the %MeHg for sediments and water was fairly consistent between the three scales of studies, varying by a factor of 6 or less. In comparison, the heavily influenced Red Devil Creek had THg values two orders of magnitude higher than the continental dataset and MeHg values less than an order of magnitude higher than the continental dataset and lower than the Crooked Creek data. This result indicates that a significant point source, such as a mercury mine, can

increase MeHg concentrations much more slowly than THg even when sulfate concentrations are higher near the mine (e.g., Table 4 in Wang 1999).

In summary, the Project baseline %MeHg of sediments and surface water is similar; if not slightly lower, than the soil %MeHg within the study area. Sediment %MeHg is also similar to other studies in the region. ProUCL-generated mean concentrations of THg and MeHg in sediments (see Section 5.3.4) also suggest an overall, low mean %MeHg of about 0.3%.

TABLE 5.1-4. SUMMARY STATISTICS FOR SURFACE WATER AT CATEGORY I SAMPLE LOCATIONS

	Min	Median	Max	Count >DL	SD	DL
THg (ng/L)	0.50	2.05	12.50	31	3.551	0.5
MeHg (ng/L)	0.027	0.062	0.093	14	0.0191	0.02
MeHg/THg (%)	0.6%	1.7%	3.9%	14	0.011	NA
pH	6.80	7.20	7.70	30	0.25	NA
Sulfate (mg/L)	1.80	10.80	53.20	30	12	0.031
%MeHg/Sulfate	0.00025	0.00225	0.00439	31	0.00119	NA
THg/Sulfate	$2*10^{-08}$	$2.2*10^{-07}$	$6.94*10^{-06}$	30	$1.3*10^{-06}$	NA
MeHg/Sulfate	$1*10^{-09}$	$6*10^{-09}$	$3.8*10^{-08}$	14	$9*10^{-09}$	NA

NA = *not applicable because the value cannot be calculated.*

Based on surface water data collected between 2013 and 2015.

TABLE 5.1-5. SUMMARY STATISTICS FOR SURFACE WATER AT CATEGORY II AND III SAMPLE LOCATIONS

	Min	Median	Max	Count >DL	SD	DL
THg (ng/L)	1.14	2.79	96.90	42	16.969	0.5
MeHg (ng/L)	0.024	0.074	0.550	29	0.1009	0.02
MeHg/THg (%)	0.2%	2.0%	3.7%	29	0.011	NA
pH	6.50	7.15	7.90	42	0.34	NA
Sulfate (mg/L)	1.51	7.56	33.20	42	5.86	0.031
%MeHg/Sulfate	0.00036	0.00202	0.00889	42	0.00181	NA
THg/Sulfate	$6*10^{-08}$	$3.5*10^{-07}$	$2.91*10^{-05}$	42	$6.6*10^{-06}$	NA
MeHg/Sulfate	$1*10^{-09}$	$8*10^{-09}$	$1.08*10^{-07}$	29	$2.7*10^{-08}$	NA

NA = *not applicable because the value cannot be calculated.*

TABLE 5.1-6. SUMMARY STATISTICS FOR SEDIMENT AT CATEGORY I SAMPLE LOCATIONS

Parameter	Min	Median	Max	Count >DL	SD	DL
THg (µg/kg)	43.4	191.0	303.0	10	73.5	0.2
MeHg (µg/kg)	0.20	0.42	0.81	10	0.2	0.01
MeHg/THg (%)	0.09	0.28	1.16	10	0.29	NA
TOC (%)	6.6	28.0	55.0	3	15.6	NA
pH	4.3	4.5	4.8	3	0.17	NA
C:N ratio calculated	15.6	19.8	80.0	3	29.4	NA
Sulfate (mg/kg)	NA	NA	NA	0	NA	10

NA= *not applicable because the value cannot be calculated.*

TABLE 5.1-7. SUMMARY STATISTICS FOR SEDIMENT AT CATEGORY II AND III SAMPLE LOCATIONS

Parameter	Min	Median	Max	Count >DL	SD	DL
THg (µg/kg)	38.0	170.0	231.0	4	80.9	0.2
MeHg (µg/kg)	0.13	0.39	1.02	4	0.34	0.01
MeHg/THg (%)	0.22	0.29	0.44	4	0.09	NA
TOC (%)	6.6	28	55	1	15.6	NA
pH	4.3	4.5	4.8	1	0.17	NA
C:N ratio calculated	NA	21.9	NA	1	NA	NA
Sulfate (mg/kg)	NA	NA	NA	0	NA	10

NA = not applicable because the value cannot be calculated.

TABLE 5.1-8. MEDIAN MERCURY CONCENTRATIONS FOR STREAM SEDIMENTS AND WATERS FROM DIFFERENT STUDIES

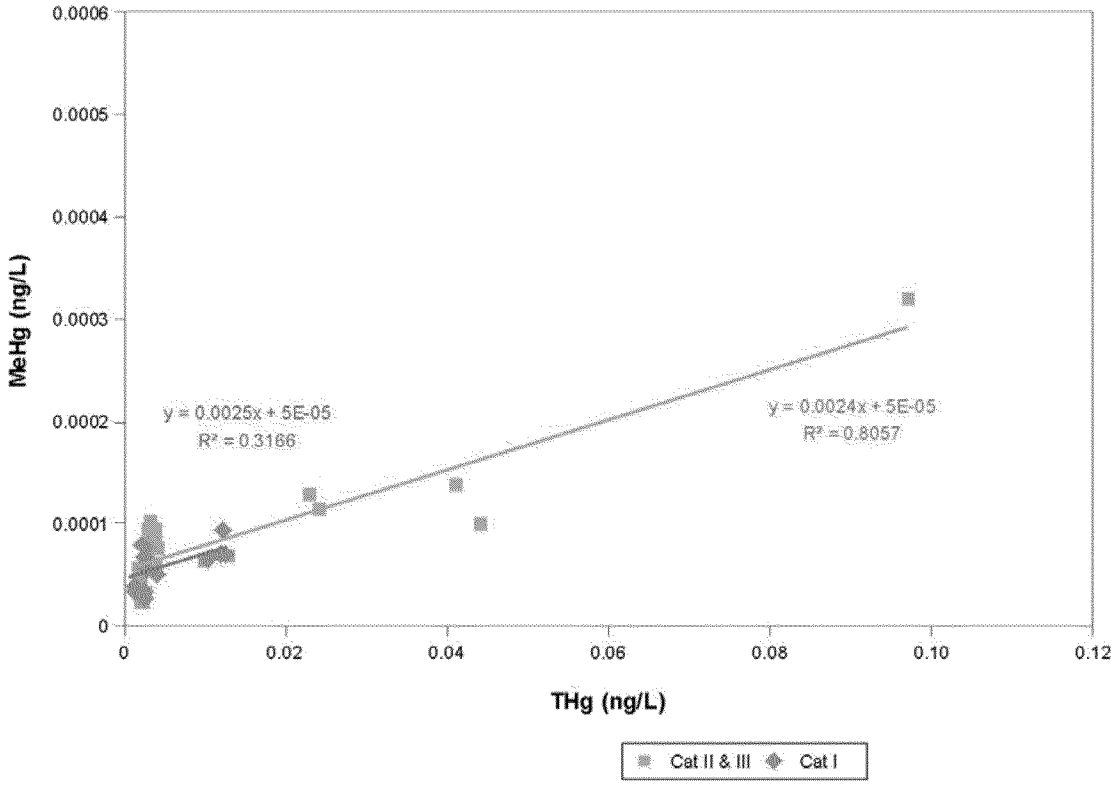
Mercury	Continental [1]	Large River Basin [2]	Small River Basin [3]
Sediment THg (µg/kg)	31.8	55.0	280
Sediment MeHg (µg/kg)	0.51	0.24	3.12
Sediment MeHg/THg (%)	1.72%	0.30%	1.11%
Water THg (ng/L)	1.90	3.80	10.5
Water MeHg (ng/L)	0.11	0.06	0.49
Water MeHg/THg (%)	5.35%	1.58%	4.67%

[1] *Median values from Scudder et al. (2009)*

[2] *Median value from Kuskokwim River and Holitna River samples from Wang (1999)*

[3] *Values from Crooked Creek from Wang (1999)*

Figure 5.1-1
THg and MeHg Results for Stream Water Quality Samples



DONLIN Proj # 0386484 | Graphics # MPF-0001-091a

5.1.4. *Review of Current Models*

As part of this assessment, existing mercury models were assessed for potential use in evaluating proposed Project effects.

In USEPA's report to Congress (1997a), USEPA noted there is a general lack of conclusive data characterizing many aspects of mercury methylation rates, and that the rate of movement of mercury from watershed soils to water bodies is prohibitive to modeling. This lack of information has caused poor predictability in modeling to date.

USEPA's report also stated that its analysis relied heavily on computer modeling to describe the environmental fate of emitted mercury because no monitoring data have been identified that conclusively demonstrate or refute a relationship between any of the individual anthropogenic sources in the emissions inventory and increased mercury concentrations in environmental media or biota. For example, USEPA (1997a) used IEM-2M to predict environmental media concentrations and the exposures that result from atmospheric mercury concentrations and deposition. The IEM-2M model is mainly soil-based and represents a watershed with untilled, upland soils (not intended for wetlands); it is also not designed to model transformations among mercury species. A significant input to the IEM-2M model was the estimate of existing mercury concentrations in the environment (including all sources of mercury). The IEM-2M has not been validated with site-specific data. The uncertainty inherent in the modeled estimates arises from many individual assumptions present within the three models. Because of these uncertainties, USEPA interpreted the model results qualitatively rather than quantitatively.

The SERAFM model was designed to simulate lakes in a watershed. The model is not designed to predict future impacts to a watershed. Rather, the model was designed to be used at contaminated sites. It is developed for application in lake systems, not wetlands, uplands, or streams. Essential data to run the model include data collected at three points within the summer with replicates, and must include:

- Filtered and unfiltered THg and MeHg from lake water column;
- THg and MeHg from lake sediments; and
- Mercury from fish tissue.

Future predictions of concentrations in media would not have the necessary data to satisfy the spatial and temporal requirements to run the SERAFM model. More importantly, the data used to develop the SERAFM model are not applicable to the wetland system being addressed in this assessment.

The D-MCM is another lake-focused model that requires speciated mercury at greater spatial and temporal frequency than the proposed Project currently has in its baseline database. Additionally, the complexity of the D-MCM model is best utilized when extensive information is known about speciated mercury in the foodweb.

Because none of the available models are applicable to the proposed Project, a calculation approach was used, as described in the following sections.

5.1.5. Future Methylation Rate Estimates

Estimates of future methylation rates considered all the environmental factors that affect methylation rates as discussed in Section 5.1.1.

5.1.5.1. Mercury Availability

Less than 10% of THg samples from baseline soil and sediment samples collected in the study area were below the detection limit, indicating that THg is not a rate-limiting parameter.

Deposition due to Donlin point and fugitive sources varies by watershed. The proposed Project is estimated to increase mercury deposition over baseline values in the nearest watersheds, decreasing with distance from the proposed Project. The average deposition rate of the affected watersheds is estimated at 0.58 micrograms per square meter per year (µg/m^2/yr) (area-weighted average of the six Crooked Creek watersheds in the study area using values from Table 4-4 in Environ (2015). This is an overestimate of the deposition rate in the study area for two reasons: (1) the model that was used to estimate deposition, CALPUFF, did not account for the pit retention of emissions; and (2) for approximately half the study area, deposition rates were estimated to be negligible (Environ 2015). Environ (2015) estimated that the deposition rate of oxidized mercury (Hg[2+]) from the proposed Project sources will be approximately 0.0116 µg/m^2/yr, or 2% of the Project deposition.

Atmospheric mercury that would be deposited would consist of Hg(0), Hg(2+), and Hg(p), with the majority in the particulate form. A minor component (approximately 1 to 2%) would be deposited as oxidized mercury (Hg[2+]), a form that is more likely to be methylated. Though Hg(2+) can easily be methylated, the rate of methylation for the newly deposited Hg(2+) is not known for the study area. As different forms of mercury may have different methylation rates, and "new" deposited mercury may be methylated more rapidly, it was assumed that newly deposited Hg(2+) (0.0116 µg/m^2/yr) would be rapidly methylated.

The global background, dominant mercury species in the atmospheric environment is gaseous elemental mercury (Hg[0]), constituting greater than 90% of total atmospheric mercury (Lyman et al. 2007). Atmospheric transport and deposition of mercury are complex, as there are chemical, diurnal, and seasonal trends that affect the redox state of mercury. Atmospheric deposition flux has been estimated globally and regionally and can vary by two or more orders of magnitude. Annual THg deposition rates from the six Mercury Deposition Network (MDN) stations located in Alaska vary between 1.2 and 5.7 µg/m^2/yr with annual THg precipitation weighted mean concentrations due to total

deposition varying between 1.5 and 10.0 ng/L. Estimated global background deposition near the proposed Project site is 8.4 µg/m^2/yr (Environ 2013).

Concentrations and deposition rates for Hg(2+) were not measured at the Alaska MDN stations. However, speciated mercury was measured at 10 sites in northern Canada (Environment and Climate Change Canada 2016) and the ratio of Hg(2+) to Hg(0) was between 0.04 and 6.64% at these sites. The two highest values (1.73% and 6.64%) were from Churchill, Manitoba and Alert, Nunavut, which are small communities located along the Arctic coastline; these sites border sea ice and are subject to atmospheric mercury deposition events (AMDEs) in spring that result in spikes of RGM and TPM not seen at the other sites. The other measurement sites and the proposed Project location are not subject to AMDEs, and, therefore, the data from Churchill and Alert were not included in this evaluation of deposition rates. The mean ratio of Hg(2+) to Hg(0) among the eight sites was 0.45%. Therefore, of the global background deposited mercury, less than 1% is Hg(2+).

5.1.5.2. Analysis of Sulfate Loading

As discussed in the above sections, the increase in methylation rate is of a similar magnitude to the increase in sulfate concentration in systems where sulfate is initially at low concentrations. The estimated sulfur dioxide (SO_2) emissions (including fugitive emissions) from the proposed Project are approximately 25.7 ton/yr. The average annual background atmospheric SO_2 concentration is approximately 1.3 µg/m^3 and annual emission will result in an additional 0.04 µg/m^3 at the Core Operating Area boundary (Rieser 2017a), which equates to a 3% increase in atmospheric SO_2. A highly conservative assumption would be that the SO_2 concentration throughout the study area increased by this same amount, and that all the SO_2 produced was deposited locally as aqueous sulfate, increasing local annual load by 3%. This would raise the average sulfate concentration in local soils and streams by approximately 3%.

Sulfate concentrations in proposed Project soil, sediment, and surface water samples were low (maximum 17.0 mg/kg in soil; see Tables 5.1-2 through 5.1-7). As many sulfate minerals are soluble and there are no Project-specific soil porewater data, the assumption was made that sulfate concentrations in the solid phase are equivalent to porewater concentrations. Compared to values reported in the literature (reviewed in Section 5.1.1.3), the proposed Project soils and sediments have sulfate concentrations that are in the methylation rate-limiting range of values. Increases to sulfate concentrations may result in increases to the methylation rate.

The reviewed research indicates that increases in sulfate concentrations are commonly accompanied by increases in MeHg concentrations of the same factor. In other words, if the proposed Project added twice the baseline sulfate load to the local receiving environment, the methylation rate would expect to approximately double. Additional increases in sulfate would not be expected to result in continued increases in

methylation rates; literature suggests that concentrations greater than 50 mg/L in aquatic systems would have diminishing returns or result in a net decrease in the methylation rate.

5.1.5.3. Other Parameters

The proposed Project is not expected to significantly impact the pH, DOC, temperature, oxygen content, or redox potential of the receiving environment. Therefore, no additional adjustments to methylation rates for items such as carbon loading were incorporated.

5.1.6. Projected Increase in Soil Mercury

Due to the absence of applicable models for evaluating potential Project effects on mercury methylation in the receiving environment, a calculation-based approach was employed. The steps of the calculation are as follows:

1. Using an average THg deposition rate of 0.582 µg/m²/yr (Environ 2015), atmospheric mercury that would be deposited would consist of Hg(0), Hg(2+), and Hg(p), with the majority in the particulate form. Oxidized mercury (Hg[2+]) is more likely to be methylated upon deposition and is a minor component (approximately 1 to 2%) of the total deposited.The rate of methylation for the newly deposited Hg(2+) is not known for the study area. As different forms of mercury may have different methylation rates, and "new" deposited mercury may be methylated more rapidly, it was assumed that all Hg(2+) was converted to MeHg. The USEPA (2005a) equation to calculate soil concentration based on deposition rate was used to calculate the future soil Hg(2+) and non-Hg(2+) concentrations from proposed Project-related atmospheric deposition. All deposited Hg(2+) was assumed to be converted to MeHg, and deposited non-Hg(2+) was assumed to undergo methylation at a rate of 1%, twice the median of all proposed Project baseline data, but reflecting the mean of paired soil samples collected in 2014 (ARCADIS 2014).

$$C_S = 100 \times \left(\frac{D}{Z_s \times BD}\right) \times t_D$$

Where:

C_s = future soil concentration (mg/kg soil)
100 = unit conversion factor (from mg-m² to kg-cm²)
D = yearly dry deposition rate of constituent (g/m²-year)
t_D = time period over which deposition occurs (years), 27 years of the Project
Z_s = soil mixing zone depth (cm); assume 2 cm
BD = soil bulk density (g/cm³); assume 1.5 g/cm³

The total increase in soil Hg(2+) and non-Hg(2+) from proposed Project-related atmospheric deposition is shown in Table 5.1-9.

2. Increases to sulfate concentration in natural systems with low concentrations of sulfate (as is the case within the study area) will result in increases in the concentration of MeHg. The reviewed literature indicates that the factor increase in MeHg is 0.5 to 1.0 times the factor increase in sulfate, or in other words, the increase in %MeHg would be of a similar magnitude to the increase in sulfate concentration in systems where sulfate is initially at low concentrations. The estimated SO_2 emissions (including fugitive emissions) from the proposed Project will result in an additional 0.04 µg/m^3, which equates to a 3% increase in atmospheric SO_2. A highly conservative assumption would be that all SO_2 produced was deposited locally as aqueous sulfate increasing local annual load by 3%. This would result in a %MeHg of existing soil THg and newly deposited non-Hg(2+) pools increasing from the current estimated methylation rate of 1% to a future rate of 1.03%.

Compounding these factors would result in MeHg soil concentrations as shown in Table 5.1-9. THg concentrations would increase by a factor of 1.01, and MeHg would increase by a factor of 1.05.

Project effects are expected to increase the THg load to the receiving environment, primarily due to deposition of Hg(p) from fugitive dust and point source emissions. Proposed Project-related emissions are also expected to increase the sulfate load to the receiving environment by approximately 3%. Increases in sulfate concentrations are commonly accompanied by increases in MeHg concentrations of the same factor. Other environmental factors commonly cited as influencing the methylation rate are not expected to be affected by the proposed Project. Consequently, the overall MeHg concentration in the study area receiving environment is expected to increase, resulting in increased future THg and MeHg soil concentrations due to both Project-related and global background deposition.

TABLE 5.1-9. CALCULATION OF FUTURE SOIL MERCURY CONCENTRATIONS

Parameter	Units	Atmospheric Deposition from Project:		Totals
		Hg(2+)	Non-Hg(2+)	
Estimated Annual Dustfall	µg/m²/yr	0.017	0.85	
Concentration in Soil (C_s)	mg/kg	0.000016	0.00077	
Future methylation rate (% of baseline)	%	---	1.03	
MeHg increase in soil due to Hg(2+) direct deposition*	mg/kg	0.0000016	---	
Mean baseline concentration, THg in soil	mg/kg			0.217
Baseline MeHg in soil at 1% methylation rate**	mg/kg			0.0022
Total Future THg in soil	mg/kg			0.218
MeHg addition to soil due to future increased methylation of future THg	mg/kg			0.0022
Total Future MeHg in soil	mg/kg			0.00226
Factor Increase, THg				1.01
Factor Increase, MeHg				1.05

**Hg will be deposited as Hg(2+) but is assumed, for purposes of this study, to rapidly convert to MeHg.*
*** An upper-bound baseline methylation rate of 1% was assumed, based on paired comparisons by ARCADIS (2014); however, ProUCL-calculated MeHg to THg means indicate an overall soil methylation rate closer to 0.5%, and sediment methylation rate <0.5%.*

5.1.7. Projected Increase in Mercury in Aquatic Systems

Mercury concentrations and %MeHg values were similar between soil and sediment datasets (sediment %MeHg was actually slightly lower than soil %MeHg), implying similar processes and rates for both systems. These comparisons indicate that methylation mechanisms are similar between soil and aquatic systems. To estimate future concentrations in aquatic systems, the same factor increase computed for soil systems was used to derive THg and MeHg in sediments. Changes in sediment mercury content should then be proportionally reflected in surface water THg and MeHg concentrations. Future sediment and surface water THg are shown in Tables 5.1-10 and 5.1-11.

TABLE 5.1-10. CURRENT BASELINE AND ESTIMATED FUTURE SEDIMENT MERCURY CONCENTRATIONS

	Minimum	Mean	95th UCLM	Maximum
Baseline THg (µg/kg)	35	179	201	303
Baseline MeHg (µg/kg)	0.057	0.503	0.627	1.6
Future THg (µg/kg)	35	179	201	303
Future MeHg (µg/kg)	0.059	0.52	0.65	1.66

TABLE 5.1-11. CURRENT BASELINE AND ESTIMATED FUTURE SURFACE WATER MERCURY CONCENTRATIONS

	Minimum	Mean	95th UCLM	Maximum
Baseline THg (ng/L)	0.55	2.76	5.66	24.30
Baseline MeHg (ng/L)	0.02	0.07	0.08	0.55
Future THg (ng/L)	0.55	2.76	5.66	24.30
Future MeHg (ng/L)	0.02	0.07	0.08	0.57

The concentration of mercury in Crooked Creek under baseflow conditions was assumed to represent mercury loading without contribution from material originating from atmospheric deposition being washed into the creek by stormwater runoff. The effect of current deposition on stream concentrations was then estimated as the increase in concentration from baseflow conditions to average annual stream concentrations.

The baseflow conditions were identified by baseflow separation analysis at four select stations where flow is continuously monitored. Concentrations for all monitoring stations during baseflow conditions were then calculated for these same time periods (TetraTech 2013). The average of the calculated baseflow THg concentrations for the stations in the Crooked Creek drainage (TetraTech 2013) was 2.6 ng/L.

Applying the factor increase of 1.01 on the baseflow THg concentrations yielded a future surface water concentration resulting from deposition from global background and proposed Project-related sources of 2.7 ng/L THg. As discussed in Section 3.4, exposure to COPCs in surface water is negligible. As described in Section 5.2.2, BAFs based on surface water concentrations are less robust than those based on sediment concentrations. Future estimated surface water concentrations are provided to show that the proposed Project will not adversely impact surface water quality.

5.2. Bioaccumulation Factors

This HHRA evaluated potential, incremental future exposures and risks related to the consumption of subsistence food items that may uptake mine-related metals. Because future concentrations of subsistence food items cannot be measured, they were estimated by developing BAFs. BAFs relate a chemical's concentration in the

environment (soil, sediment, water) to its concentration in biota and are commonly used in risk assessments to estimate exposures via plant, fish, or wildlife ingestion.

Mercury, arsenic, and antimony can bioaccumulate in biota. Mercury is known to biomagnify through the food chain, while arsenic and antimony do not; consequently, mercury concentrations tend to be higher in higher trophic level organisms. Studies have shown for example that predatory fish tend to have higher mercury levels than omnivorous or herbivorous species (De Lacerda 1994, Eisler 1987).

Because mercury can biomagnify, the form of mercury is important to understand uptake and bioaccumulation potential. While inorganic mercury does not biomagnify, MeHg is of particular concern because it can build up in certain edible fish and mammals to levels that are many times greater than levels in the surrounding environment (ATSDR 1999).

Mercury must be absorbed into the body of humans or animals, or taken up by plants to cause a toxic response. Absorption can occur through oral ingestion, dermal absorption, or inhalation. Inorganic mercury absorption following dermal, oral, or inhalation exposure is low; absorption following oral ingestion is estimated for example between 10 to 30% depending on the medium in which the mercury is ingested (e.g., gavage, water, milk, or food). Estimated absorption rates of organic mercury forms are much higher than inorganic forms, as much as 95% in humans when consumed in drinking water (Aberg et al. 1969 as cited in ATSDR 1999). Studies suggest that MeHg comprises 90% or more of the THg in fish (ATSDR 2013).

Given the differences in bioavailability between inorganic and organic mercury, as well as limitations of mercury data in biota (i.e., THg was often measured but MeHg was seldom measured), proposed Project-specific uptake factors for mercury were derived using:

- THg data for primary media and THg data for biota; and
- MeHg data for primary media and THg data for biota.

These two types of BAFs were developed to allow for comparisons to other regional data and the general literature, and to account for potential future changes in bioavailable forms of mercury in the environment. MeHg is readily bioavailable and taken up by biota, whereas the bioavailability of THg is more variable. Therefore, deriving MeHg to THg BAFs allows for some accountability of increasing proportions of MeHg in future soils.

Existing literature suggests that MeHg is likely to comprise a relatively large proportion of the THg that is measured in biota (ATSDR 1999, 2013). For example, as noted above, ATSDR (2013) reports that MeHg comprises 90% or more of the THg in fish. However, not all studies indicate this is the case; Gray et al. (2000) reported that MeHg comprised less than 1% of the THg in aquatic biota collected at three mine sites in southwestern Alaska. However, given the variability reported in the literature of the fraction of THg that is MeHg (less than 1% to over 90%) and the relative toxicity of MeHg to THg

(ATSDR 1999), the assumption in this HHRA was that all THg in biota is MeHg. This assumption results in an overestimate of potential risk from consumption of subsistence foods due to exposures to mercury.

Baseline biota values were estimated using BAFs developed from Project-specific data or BAFs from the literature, if Project-specific data are lacking. Appendix B describes in detail how BAFs were developed or selected for mercury, arsenic, and antimony for representative subsistence foods, which included berries (blueberry and cranberry), northern pike, mallard ducks, and beaver.

Deriving bioaccumulation models using empirical datasets is preferred to using literature-based (generic) models when estimating concentrations in food items (Sample et al. 1998, Bechtel 1998). Additionally, due to the heterogeneity of bioaccumulation patterns in environmental media, proposed Project-specific empirical data are preferred over other, regional data (or models). Use of regional data or models is less likely to be representative of site conditions and is likely to result in an unspecified level of uncertainty when estimating exposure.

Though proposed Project-specific empirical data could be used to develop BAFs, there were limited Project data to develop regression model-based BAFs. Therefore, in most cases, median concentrations of primary media and biota tissue were used to derive BAFs. Compared to other literature or regional information (Appendix B), the proposed Project-specific BAFs were high; suggesting BAFs for the study area may be overestimated. Reasons for the higher Project-specific BAFs are not known, but may in part be due to the nature of the sampling events during baseline collection, which sought to characterize the more mineralized areas of the study area. These more mineralized areas may not be representative of the study area as a whole. However, the bias represents a conservative assumption. The derivation of the BAFs for use in the HHRA is described in detail in Appendix B. Table 5.2-1 provides the BAFs that were used in the HHRA.

TABLE 5.2-1. BIOACCUMULATION FACTORS USED IN THE HUMAN HEALTH RISK ASSESSMENT

Food	Media	Mercury [1]	Arsenic	Antimony
Plants – berries	Soil	2.9	0.0375	$\ln(C_{plant}) = 0.938 \times \ln(C_{soil}) - 3.233$
Fish – northern pike[2]	Sediment	55	0.016	0.018
Bird – mallard	Soil	0.054	$\ln(C_{bird}) = 0.8188 \times \ln(C_{soil}) - 4.8471$	0.05
Mammal – beaver	Soil	0.054	$\ln(C_{mammal}) = 0.8188 \times \ln(C_{soil}) - 4.8471$	0.05

The soil-to-plant uptake factor for mercury is in units of $kg_{soil,dw}/kg_{tiss,ww}$.

The soil-to-plant uptake factors for arsenic and antimony are in units of $kg_{soil,dw}/kg_{tiss,dw}$.

All sediment-to-fish uptake factors are in units of $kg_{sed,dw}/kg_{tiss,ww}$.

All soil-to-mammal uptake factors are in units of $kg_{soil,dw}/kg_{tiss,dw}$.

[1] *Uptake factors (median) are based on Project-specific MeHg data for soils and THg data for biota.*

[2] *Uptake factor modeled using sculpin data. Food chain multiplier of 3 is applied to this value to derive northern pike concentrations.*

5.2.1. *Bioaccumulation in Berries*

Project-specific, baseline soil and berry tissue data were used to develop soil-to-plant uptake factors to use for future estimates of uptake of THg into blueberries and cranberries ("berries"). The datasets included site-specific baseline sampling for the proposed Project, which included a total of 137 berry samples (71 blueberry samples and 66 cranberry samples) collected from 43 locations in 2006 and 2007. The specific parts of the plants that were collected were not reported; however, the stated objective of the sampling program was to collect data relevant to subsistence harvesting (ARCADIS 2007a), and it was therefore assumed that the data represent berry data that would be consumed by humans. Project-specific data for antimony and arsenic were not available. Literature-based BAFs were used for antimony and arsenic.

5.2.2. *Bioaccumulation in Fish*

Northern pike is a resident, large, and relatively long-lived top predator. For these reasons, it provides a conservative estimate of potential fish uptake of metals, particularly mercury. However, northern pike has only recently been observed in Crooked Creek (OtterTail 2009). In addition, the Project-specific tissue burden dataset for northern pike consists of only two samples, only one of which was measured in an adult individual, and only THg was analyzed for tissue burden. Given the limited dataset, other fish species were considered for the HHRA.

A fairly large, Project-specific sculpin dataset was available for the study area. Although sculpin are not large, top predators and are not widely harvested for subsistence consumption, other regional studies have used sculpin data to estimate mercury concentrations in higher trophic level game fish such as northern pike (E&E 2014). As discussed in Appendix B, sculpin BAFs were developed using surface water and sediment data. The sediment-based BAFs are considered more robust (see Appendix B) and recommended for use in estimating fish concentrations for the HHRA. Due to the greater bioavailability of MeHg in sediment, the MeHg to THg BAF was used. Sculpin are one trophic level lower than northern pike, and to account for greater potential bioaccumulation of MeHg in northern pike, a food chain multiplier (FCM) of 3 was applied to sculpin MeHg concentrations derived using the sediment to fish BAF (McGeer et al. 2004; E&E 2014). An FCM of 1 was applied to arsenic and antimony, which do not biomagnify through the food chain.

5.2.3. *Bioaccumulation in Beaver*

There is no Project-specific or regionally available empirical data that relate metal concentrations in the environment to tissue burdens of wild mammal game. However, there are models available that characterize the bioaccumulation of metals into mammals. Thus, given the lack of Project-specific data, literature-based soil-to-mammal uptake factors from USEPA (2005b) for arsenic, USEPA (2005c) for antimony, and Sample et al. (1998) for mercury were used to estimate tissue burdens in beaver.

5.2.4. *Bioaccumulation in Mallard Ducks*

There are no Project-specific empirical data that relate metals concentrations in the environment to tissue burdens of birds. Similarly, few data exist in the peer-reviewed literature or regulatory guidance that relate metals concentrations in the environment to tissue burdens of birds. While there is a large amount of literature documenting the accumulation of metals in birds, the majority of these studies often do not include both co-located environmental and bird tissue concentrations and, hence, cannot be used to develop quantitative uptake models. Therefore, available uptake factors for small mammals were adopted for birds.

5.3. Baseline Exposure Point Concentrations

To calculate health risks, an estimate must be made of the constituent concentration to which an individual may be exposed. An EPC is the concentration of a COPC in an exposure medium at the location where a receptor may contact that medium, and that is representative of the time period over which exposure may occur.

For all media except air, the USEPA guidance document to calculate EPCs (USEPA 2002), and recommendations in the associated software program ProUCL, were the primary references used to calculate EPCs. USEPA's *ProUCL v5.1* software (USEPA 2016a) was used. Empirical data were not available for all COPCs in small mammals and waterfowl, and for arsenic and antimony in berries; EPCs were estimated for these media using BAFs. Baseline mercury air concentrations were based on analyses completed by Environ (2015). Baseline antimony and arsenic air concentrations were assumed to be negligible, and for calculation purposes were assumed to be zero.

USEPA (1992a, 2002) guidance recommends using an upper-bound estimate of the average concentration to which an individual would be exposed over a significant part of a lifetime, due to the uncertainty associated with estimating the true average concentration at a site. Upper-bound and mean estimates of baseline EPCs were computed; however, future EPCs are based on modeling estimates, which generate mean estimates. Therefore, mean baseline EPCs were used to estimate baseline risks to evaluate incremental risk.

The data distributions encountered while computing the EPCs fell into one of three categories: normal (having a symmetric, bell-shaped distribution about the mean), lognormal with a small coefficient of variation (where the distribution of values around the mean is somewhat skewed towards larger values), and neither normal nor lognormal. The Shapiro-Wilk W-test is a statistical hypothesis test used to determine whether the data are normally or lognormally distributed, or consistent with neither distribution. A significance level of 0.05 was employed in the Shapiro-Wilk W-test.

Table 5.3-1 presents the baseline EPCs.

TABLE 5.3-1. BASELINE EXPOSURE POINT CONCENTRATIONS OF CONSTITUENTS OF POTENTIAL CONCERN IN ENVIRONMENTAL MEDIA

COPC	Baseline Exposure Point Concentration						
	Air (µg/m³)	Soil (mg/kg dw)	Sediment (mg/kg)	Berries (mg/kg ww)	Northern Pike (mg/kg ww)	Beaver (mg/kg ww)	Mallard Duck (mg/kg ww)
Antimony	NA	4.93	0.3	0.0539	0.00610	0.0789	0.0813
Arsenic	NA	15.1	24.4	0.173	0.390	0.0232	0.0239
Mercury	0.0000084	0.217	0.179	NA	NA	NA	NA
MeHg	NA	0.00087	NA	0.00254	0.0829	0.00375	0.00387

COPC = constituent of potential concern

dw = dry weight

ww = wet weight

NA = Not applicable (mercury in biota is assumed to be 100% MeHg; antimony and arsenic are negligible in baseline air)

The baseline exposure point concentrations are mean values.

Arsenic concentrations in biota were converted to inorganic arsenic in dose calculations.

5.3.1. *Air*

Baseline air quality is described in the DEIS, Chapter 3, Section 3.4. Details of the baseline sampling program can be found in Environ (2013).

Because of the lack of Project-specific or regional data for antimony and arsenic concentrations in air, the estimated incremental increase in concentrations was evaluated with the assumption that baseline air concentrations of these COPCs were zero. This is a conservative assumption, as these COPCs are present in soil in the region and particulates in air under baseline conditions would be expected to contain minor amounts of these COPCs.

To determine inhalation EPCs for mercury, it is necessary to calculate the baseline COPC concentrations bound to fine particulate matter, as it is the fine size fraction of particles that can be inhaled deep into the alveoli. Furthermore, mercury bound to particulate matter is primarily present in the fine fraction of particulate matter ($PM_{2.5}$), although some particulate-bound mercury may also be present in coarse particulate matter (Bullock and Brehme 2002).

A baseline annual average mercury concentration bound to $PM_{2.5}$ (0.0000084 $\mu g/m^3$) was obtained from Environ (2013). This concentration was based on monitoring data collected in the study area. Baseline mercury air concentrations are not expected to vary widely in the area in general due to the large global background of atmospheric mercury (Environ 2015). Therefore, the spatial representativeness was determined to be acceptable for the HHRA. Because methylation does not occur in the atmosphere, it was assumed that all atmospheric mercury is inorganic for risk evaluation.

5.3.2. *Soil*

Soil datasets considered for baseline EPCs, and also used in BAF calculations, included site-specific baseline sampling for the proposed Project (ARCADIS 2007a, 2014; Fernandez 2014), which included 122 soil samples from 45 locations, collected between 2006 and 2013. The samples were collected within the Crooked Creek Hydrologic Unit Code (HUC) 10 drainage. Soil arsenic within the Core Operating Area is locally enriched relative to the samples collected outside this area (Fernandez 2014); because exposures will occur outside the Core Operating Area, soil arsenic data was excluded from the data used to determine baseline soil EPCs. All soil arsenic data, however, was used to determine potential BAFs and air concentrations. Samples were analyzed for THg (n = 52; all samples were above the detection limit [DL]) and MeHg (n = 20; eight samples were below the DL); total arsenic (n = 35; all samples were above the DL); and total antimony (n = 35; all samples were above the DL). It was assumed that 100% of the total arsenic in soil was inorganic.

The baseline distributions of antimony and arsenic concentrations in soil were not normal (at a 5% significance level) and did not follow a discernable distribution. The

baseline distribution of mercury concentrations in soil was lognormal (at a 5% significance level) and the baseline distribution of MeHg concentrations in soil was normal (at a 5% significance level).

5.3.3. *Berries*

Blueberry and cranberry ("berry") Project-specific datasets (ARCADIS 2007a) were used to generate BAFs (Section 5.2.1).

Baseline concentrations of antimony, arsenic, and mercury were estimated using BAFs for comparability to future concentrations.

5.3.4. *Small Mammals (Beaver)*

Baseline beaver tissue metals concentrations were derived from small mammal BAFs. The arsenic concentration in beaver was adjusted to account for the amount of inorganic arsenic that is likely to be present in tissues. Inorganic arsenic is the most toxic form of arsenic; also the oral reference dose (RfD) for arsenic is based on inorganic arsenic (USEPA 2017). For beaver, it was assumed that 70% of the total arsenic was inorganic (European Food Safety Authority [EFSA] 2009, 2014).

5.3.5. *Waterfowl (Mallard Duck)*

Baseline mallard duck tissue metal concentrations were derived from mallard duck BAFs (small mammal BAFs were used due to lack of BAFs for avian species). For mallard duck, it was assumed that 50% of the total arsenic was inorganic (EFSA 2009, 2014).

5.3.6. *Fish*

In the HHRA scope of work (ERM 2017), northern pike were identified to be a representative resident species that would be used to estimate incremental risks to humans from fish consumption. However, as discussed in Section 5.2.2, data were only available for two northern pike tissue samples, one from an adult individual, and one from a juvenile individual. Both samples were analyzed for THg (in wet weight [ww]). The adult northern pike tissue THg concentration was 0.421 mg/kg ww and the juvenile northern pike tissue THg concentration was 0.0852 mg/kg ww.

Both the small sample size and the age and trophic level disparity in the two samples (juvenile northern pike consume plants, while adult northern pike consume fish and are more commonly consumed) created relatively large uncertainties with the use of this dataset. To fulfill the HHRA scope, risks were estimated using northern pike concentrations modeled from sculpin BAFs and a FCM of 3 to account for the higher trophic level of northern pike.

For fish tissue, mercury is assumed to be present as 100% MeHg (ATSDR 1999). Based on studies in various fish species (Phillips 1990, Slejovec et al. 2004, Rosemond et al. 2008, and Rahman et al. 2002), it is assumed that 10% of the arsenic is inorganic. Organic forms of arsenic have a low to negligible toxicity to humans.

5.4. Future Media Concentrations

Future EPCs were estimated by modeling. While there are always inherent uncertainties with estimating future concentrations, in general, simple models with conservative assumptions are favored over complex models that bring additional uncertainties to future EPC estimation. However, sensitivity runs with a few key parameters were made and are presented in the Uncertainty Analysis (Section 8) to examine HHRA outcomes with respect to incremental risks. The subsections below describe how future EPCs were computed. Table 5.4-1 presents a summary of all the future EPCs.

TABLE 5.4-1. FUTURE EXPOSURE POINT CONCENTRATIONS OF COPCs IN ENVIRONMENTAL MEDIA

COPC	Future Exposure Point Concentration						
	Air (µg/m³)	Soil (mg/kg dw)	Sediment (mg/kg)	Berries (mg/kg ww)	Northern Pike (mg/kg ww)	Beaver (mg/kg ww)	Mallard Duck (mg/kg ww)
Antimony	5.71E-07	4.94	0.3	0.0539	0.00611	0.0790	0.0814
Arsenic	1.80E-05	15.3	24.4	0.175	0.391	0.0234	0.0241
Mercury	8.46E-6	0.219	0.181	NA	NA	NA	NA
MeHg	NA	0.00092	0.000528	0.00266	0.0871	0.00379	0.00391

COPC = constituent of potential concern

dw = dry weight

ww = wet weight

NA = Not applicable due to assumption that mercury in biota is MeHg and MeHg is not present in air.

Arsenic concentrations in biota were converted to inorganic arsenic in dose calculations.

5.4.1. *Air*

The air quality model developed to support the Prevention of Significant Deterioration (PSD) permit application (Air Sciences 2016, Donlin 2016) was used to simulate the incremental annual average $PM_{2.5}$ concentration at the village of Crooked Creek that would occur as a result of activity at the mine. The air quality model simulated impacts from the mine during the projected peak emission year, which is mine year 16. Several receptor locations were selected around the village, and the model was run using five years of meteorological data collected at the site. The location with the highest average $PM_{2.5}$ concentration was used to characterize the $PM_{2.5}$ material originating from the proposed Project. The antimony, arsenic, and mercury concentrations in the $PM_{2.5}$ material were then calculated using the relative concentrations of these metals in dust particulate material from the site emissions inventory prepared for the PSD permit application (Donlin 2016, Air Sciences 2017).

As reported by Air Sciences (2017), the predicted annual average mercury concentration was a maximum of 0.00148 µg/m^3 within the Crooked Creek village watershed. This concentration was used to calculate risk to residents via inhalation. The concentration was multiplied by the weighted percentage of particulate to THg emissions (2.8%; Environ 2015) to obtain the predicted annual average particle-bound mercury concentration (0.0000414 µg/m^3). The predicted annual average particle-bound antimony and arsenic concentrations were calculated in the same manner.

5.4.2. *Soil*

Soil concentrations of COPCs are estimated to increase over the life of the mine due to atmospheric deposition of COPCs emitted from mine operations. The USEPA method for calculating constituent concentrations in soil due to atmospheric dust deposition was used to estimate future soil concentrations (USEPA 2005a). Future dustfall levels were based on air quality estimates. As reported by Environ (2015), the simulated maximum average deposition flux of Hg(p) due to proposed Project-related sources in local watersheds ranged from 0.1 to 3.7 micrograms per square meter per year (µg/m^2/yr). An area-weighted average value (0.582 µg/m^2/yr) was adopted as the annual dustfall deposition for mercury, to be used for predicting future soil mercury concentrations.

Arsenic and antimony deposition rates were estimated by using the concentration of particulate arsenic and antimony relative to mercury as calculated for the PSD application (Donlin 2016), and multiplying the mercury deposition attributable to fugitive particulates (Environ 2015) by these ratios.

For soil quality modeling, in addition to assumptions made in the air dispersion model (Environ 2015), it was conservatively assumed that all dust deposited onto soil remains in place and would not be entrained in run-off during rain events. Further, it was

assumed that the predicted concentrations of COPCs in dustfall apply to the entire study area.

The incremental increase in soil COPC concentrations due to dustfall deposition was calculated for each COPC using Equation 1, as suggested by the USEPA (2005a):

$$C_S = 100 \times \left(\frac{D}{Z_s \times BD}\right) \times t_D \qquad \text{[Equation 1]}$$

where:

C_s = average soil concentration over exposure duration (mg COPC/kg soil)
100 = unit conversion factor (from mg-m^2 to kg-cm^2)
D = yearly dry deposition rate of constituent (g COPC/m^2-year)
t_D = time period over which deposition occurs (years)
Z_s = soil mixing zone depth (cm)
BD = soil bulk density (g/cm^3)

The time period (t_D) over which dust deposition may occur was assumed to be 27 years of proposed Project operations. The COPCs deposited with fugitive dust were assumed to mix with the top 2 cm of soil (Z_s), as recommended by the USEPA (2005a) for untilled soils. The bulk density (BD) for soil was set at the default value of 1.5 g soil/cm^3 soil, as recommended by the USEPA (2005a). Weathering and degradation are considered to only be significant for organic constituents and not metals (USEPA 2005a); thus, a soil loss constant was not necessary (i.e., it was assumed that none of the metals were lost to weathering or degradation).

The incremental increase in soil COPC concentrations was summed with baseline COPC concentrations to estimate total future soil COPC concentrations in the study area.

5.4.3. *Sediment*

Future dustfall containing arsenic and antimony was assumed to be deposited in stream sediments. Sediment antimony and arsenic concentrations were therefore modeled to increase proportional to dustfall. Sediment mercury concentrations were based on analyses presented in Section 5.1.7.

5.4.4. *Berries*

Predicted berry concentrations were calculated by multiplying the predicted future soil concentration by the MeHg (soil) to THg BAF for berries. It was assumed that 100% of THg in berries is in the form of MeHg. Although dustfall onto berries is also a factor in the total berry concentration, for the purposes of the HHRA, the soil to berry uptake route was assumed to be the primary exposure route to berries. Uncertainties with regard to additional exposure via dustfall onto berries are addressed in the Uncertainty Analysis (Section 8).

5.4.5. *Small Mammals (Beaver)*

Future beaver tissue metals concentrations were calculated from predicted soil concentrations and the MeHg (soil) to THg BAF (see Section 5.2.4). Note that future mercury concentrations in beaver were predicted using the MeHg content in soil to obtain the THg concentration in beaver tissue. The same fractions of total arsenic to inorganic arsenic, and THg to MeHg, in beaver were assumed for future EPCs as for baseline EPCs.

5.4.6. *Waterfowl (Mallard Duck)*

Future mallard tissue metal concentrations were calculated from predicted soil concentrations and BAFs (see Section 5.2.3). Note that future mercury concentrations in beaver were predicted using the MeHg content in soil to obtain the THg concentration in mallard duck tissue. The same fractions of total arsenic to inorganic arsenic, and THg to MeHg, in mallard ducks, were assumed for future EPCs as for baseline EPCs.

5.4.7. *Fish*

Predicted northern pike tissue concentrations were calculated by multiplying the predicted future sediment concentrations by the BAF (based on MeHg in sediment) for sculpin. Details are presented in Section 5.2.2 and Appendix B. It is assumed that 100% of the mercury in fish is MeHg. Ten percent of the estimated arsenic concentrations were assumed to be inorganic arsenic, as was assumed for baseline EPCs.

5.5. Intake Rates

Exposure factors define the magnitude, frequency, and duration of exposure for the populations and pathways selected for quantitative evaluation. Exposure factors are combined with EPCs to calculate dose. Exposure factors were selected based on a "reasonable maximum exposure" scenario that combines upper-bound and average values that reflect exposures somewhere between the 90th and 98th percentile of the range of possible exposures that reasonably can be expected to occur for a given population (USEPA 1989).

The basic structure of the exposure equations used in this HHRA is from *Risk Assessment Guidance for Superfund, Human Health Evaluation Manual, Part A* (USEPA 1989). The general intake equation that serves as the basis for the pathway-specific equations is:

$$Intake = \frac{C_i \times CR \times EF \times ED}{BW \times AT} \qquad \text{[Equation 2]}$$

where:

Intake = rate of COPC available for uptake at an exchange boundary (mg /kg body weight/day)

C_i = concentration of constituent *i* at exposure point (e.g., mg/kg sediment)

CR = contact rate with the environmental medium (e.g., mg sediment/day)

EF = exposure frequency (days/year)

ED = exposure duration (year)

BW = body weight (kg)

AT = averaging time for toxicological effects (days)

Intake is specified as the rate at which a constituent becomes available for uptake at an exchange boundary, such as the walls of the gastrointestinal tract or the skin. Hence, it is not equivalent to an absorbed dose, which is the amount of a constituent actually entering the bloodstream across an exchange boundary.

5.5.1. *Human Receptor Characteristics*

The human receptors selected were children (0 to 6 years old) and adults (greater than 20 years of age). Children are often most susceptible to constituents with a threshold response due to their ratio of body size to ingestion rates (IRs) compared to other human life stages.

Health effects are generally divided into two categories: threshold (non-carcinogenic) and non-threshold (carcinogenic) responses. Constituents may elicit one or both types of response. Because these two categories of constituents are evaluated differently, separate intake calculations are performed. For non-carcinogenic effects, the averaging time over which effects are assessed is equal to the exposure duration (USEPA 1989), and the resulting potential for effects is the ratio of the estimated dose to a reference dose. Cancer risk, on the other hand, is assessed as exposure averaged over a lifetime, and is expressed as a probability of incremental lifetime cancer risk (additional cancer incidences in the population to the background cancer risk).

Consumption (i.e., ingestion) rates of food items were based on a local subsistence study that reported data for the Upper Kuskokwim River subsistence population, as described in Sections 3.6 and 5.5.2.3. For other intake rates, ADEC (2015) or USEPA (2014) exposure factors for residential receptors were selected.

The receptor characteristics for adults and children used to calculate the dose of COPCs from the different exposure pathways are summarized in Table 5.5-1. Pathway-specific intake equations are described in the following subsections.

TABLE 5.5-1. EXPOSURE FACTORS FOR ADULT AND CHILD SUBSISTENCE POPULATIONS

			Adult		Child [2]	
Exposure Parameter [1]	Symbol	Units	Value	Reference	Value	Reference
Body Weight	BW	kg	80	10	15	10
Exposure Duration	ED	years	21	4	6	4
Exposure Frequency - Soil [11]	EFsoil	days/year	270	10	270	10
Exposure Frequency - Subsistence Foods	EFfoods	days/year	365	10	365	10
Averaging Time (non-carcinogens)	ATnon-carcinogens	days	7,665	4	2,190	4
Averaging Time (carcinogens)	ATcarcinogens	days	25,550	10	25,550	10
Relative Bioavailability (general)	RBA	unitless	1	8	1	8
Relative Bioavailability (arsenic)	RBAAs	unitless	0.6	9	0.6	9
Ingestion Rates						
Soil - Incidental Ingestion	IRs	kg/day	0.0001	5	0.0002	5, 10
Subsistence Foods - Beaver	IRbeaver	kg/day	0.0129	6	0.00841	7
Subsistence Foods - Mallard Duck	IRduck	kg/day	0.00113	6	0.000740	7
Subsistence Foods – Pike	IRpike	kg/day	0.00909	6	0.00593	7
Subsistence Foods - Berries	IRberry	kg/day	0.0148	6	0.00968	7
Inhalation Factors						
Exposure Time	ET	hours/day	24	4	24	4
Exposure Frequency	EF	days/year	365	4	365	4
Exposure Duration [12]	ED	years	21	10	6	10
Averaging Time (non-carcinogens)	ATnon-carcinogens	hours (ED [years] x 365 days/year x 24 hours/day)	183,960	3	52,560	3
Averaging Time (carcinogens)	ATcarcinogens	hours (Lifetime [years] x 365 days/year x 24 hours/day)	613,200	10	613,200	10

Notes:

[1] The exposure assumptions presented are used to assess non-cancer and cancer risk. Default exposure parameters were selected preferentially from ADEC (2015) and then USEPA (2014). Parameters not available in these sources were selected from other agency guidance or peer-reviewed literature. Reference for each value is identified in the table and defined below.

[2] Assumed children were in the age group of 0 to 6 years old (USEPA 2011).

[3] USEPA (2011).

[4] Site-specific characteristics.

[5] USEPA (2014).

[6] Brown et al. (2012).

[7] Consumption rates for children are assumed to be 65% of adult consumption rates.

[8] USEPA (2007b).

[9] USEPA (2012).

[10] ADEC (2015).

[11] The soil exposure frequency is based on the climate zone of the proposed Project. Subsistence population soil exposure frequency is 270 days/year for the under 40-inch zone, in which the proposed Project is located.

[12] ADEC (2015) recommends an exposure duration for adults of 20 years and for children 6 years. However, because the Project Operations phase lasts 27 years, the exposure duration for adults was assumed to be 21 years, for a total of 27 years when adult and child exposure durations are summed.

5.5.2. *Non-Carcinogenic Exposure*

5.5.2.1. Exposure via Air Inhalation

Following USEPA (2009a) guidance, risk due to air exposure requires a calculation of the exposure concentration.

The equation for the exposure concentration at the point of exposure (µg/m^3) from inhalation of COPCs bound to $PM_{2.5}$ is (USEPA 2009a):

$$EC = \frac{C_A \times ET \times EF \times ED}{AT} \quad \text{[Equation 3]}$$

where:

EC = exposure concentration of COPC in air (µg/m^3)
C_{Air} = concentration of COPC in air (µg/m^3)
ET = exposure time (hours/day)
EF = exposure frequency (days/year)
ED = exposure duration (years)
AT = averaging time (ED in years x 365 days/year x 24 hours/day)

5.5.2.2. Exposure via Soil Ingestion

The equation used to calculate the average daily dose (ADD) of COPCs (mg/kg BW/day) from ingestion of soil is (USEPA 1992b):

$$ADD = \frac{C_S \times IR_S \times EF \times ED}{BW \times AT} \quad \text{[Equation 4]}$$

where:

C_S = concentration of COPC in soil (mg/kg)
IR_S = soil ingestion rate (kg/day)
EF = exposure frequency (days/year)
ED = exposure duration (days)
BW = body weight (kg)
AT = averaging time (days)

The assumptions used in the calculation of the ADD of COPCs via soil ingestion are as follows:

- Baseline soil quality at the sampling sites is representative of baseline soil quality within the study area;
- Adults and children are exposed 270 days per year, the value recommended by ADEC (2015) for exposure in the under 40-inch zone;
- Children have a soil IR of 0.0002 kg/day and a body weight (BW) of 15 kg (ADEC 2015); and

- Adults have a soil IR of 0.0001 kg/day and a BW of 80 kg (ADEC 2015).

5.5.2.3. Exposure via Subsistence Food Ingestion

The consumption rates of subsistence foods were based on per capita harvest rates (kg/year) of subsistence foods (i.e., northern pike, beaver, mallard duck, and berries) for eight communities in the Upper Kuskokwim group (Brown et al. 2012). These communities include Aniak, Chuathbaluk, Crooked Creek, Lower Kalskag, Red Devil, Sleetmute, Stony River, and Upper Kalskag. Because no information was provided to estimate the proportion of harvested food that was subsequently consumed, it was assumed, for the purposes of the HHRA, that all harvested food items were consumed. The use of harvest rates to estimate consumption rates is an overestimate; the study indicated that people do not consume all harvested food, and further, people may not necessarily consume the total amount of harvested food components (e.g., often just the fish fillet is consumed, not the entire fish). However, by utilizing the harvested amount to estimate intake, this HHRA overestimates potential exposures from this pathway: if no incremental risk is indicated by this approach, then groups ingesting only a portion of harvested foods are unlikely to be at risk.

Equation 4 was also used to calculate the ADD for COPCs except arsenic (mg/kg BW/day) from the ingestion of subsistence foods, by using the tissue concentration in place of the soil concentration and the ingestion rate for the specific food in place of the soil ingestion rate. To calculate the ADD for arsenic (mg/kg BW/day) from the ingestion of subsistence foods, the following equation was used:

$$ADD = \frac{(C_F \times IAF \times IR_F) \times EF \times ED}{BW \times AT} \qquad \text{[Equation 5]}$$

Where:

C_F = concentration of COPC in subsistence food (mg/kg)
IAF = Inorganic arsenic fraction in subsistence food (%)
IR_F = ingestion rate of subsistence foods (kg/day)
EF = exposure frequency (days/year)
ED = exposure duration (days)
BW = body weight (kg)
AT = averaging time (days)

The subsistence food consumption rates for children (0 to 6 years of age) were assumed to be 65% of an adult's rate, based on comparison of the overall food IR of adults to overall food IR of children. The total mean consumption rate from the mean per capita food intakes and body weights presented in the USEPA Exposure Factors Handbook (USEPA 2011) for adults was compared to those for children. For an adult, average total food consumption is 2,320 g/day (29 g/kg/day for an 80 kg adult). For children 1 to <3 years old, the average daily total food consumption is 1,559 g/day (113 g/kg-day and

body weight 13.8 kg). For children 3 to <6 years old, the total food consumption is 1,469 g/day (79 g/kg-day and body weight 18.6 kg). The average of the two child IRs is 1,514 g/day. Dividing the mean total consumption rate for the two child categories (1 to <3 years old and 3 to <6 years old) by the total mean consumption rate for the adult indicates that children consume at a rate that is 65% of the adult rate. This factor is applied to the adult consumption rate for individual subsistence foods to obtain the corresponding child consumption rate.

5.5.3. Carcinogen Exposure

5.5.3.1. Inhalation

Arsenic is considered to be a carcinogen via the inhalation exposure route; however, baseline concentrations of arsenic bound to $PM_{2.5}$ were assumed to be zero (Section 5.3.1), thus there is no exposure to arsenic via inhalation under baseline conditions and no carcinogenic risk.

Under future conditions, the predicted annual average arsenic concentration bound to $PM_{2.5}$ (which is equivalent to the concentration in air) is 0.000018 µg/m^3. Equation 3 was modified for carcinogens to use lifetime for the averaging time, in hours.

5.5.3.2. Ingestion

The exposure to carcinogens via ingestion was calculated with the lifetime average daily dose (LADD), using the following equation (USEPA 1992a):

$$LADD = \frac{C \times IR \times EF \times ED}{BW \times AT} \qquad \text{[Equation 6]}$$

Where:

C	= concentration of the COPC (mg/kg)
IR	= ingestion rate (kg/day)
EF	= exposure frequency (days/year)
ED	= exposure duration (days)
BW	= body weight (kg)
AT	= averaging time (for carcinogens, AT is lifetime) (days)

The exposure duration for the site was assumed to be 21 years for an adult and 6 years for a child (7,665 and 2,190 days, respectively). Receptor lifetime is assumed to be 70 years (ADEC 2015), which is equivalent to 25,550 days.

The concentrations of arsenic in the subsistence food items were adjusted to account for the amount of inorganic arsenic (see Section 5.3), which was used in the calculation of the LADD for subsistence foods.

5.6. Dose and Concentration Estimates

The baseline inhalation exposure concentrations and ingestion doses from all pathways for adult and child human receptors are shown in Tables 5.6-1 and 5.6-2. Tables 5.6-3 and 5.6-4 show the future inhalation exposure concentrations and ingestion doses from all pathways for adult and child human receptors.

TABLE 5.6-1: BASELINE DOSE OF CONSTITUENTS OF POTENTIAL CONCERN FOR ADULT SUBSISTENCE

COPC	Oral - Soil ADD	Oral - Beaver ADD	Oral - Mallard Duck ADD	Oral - Pike ADD	Oral - Berries ADD	Inhalation EC
Antimony	4.56E-06	1.27E-05	1.15E-06	6.93E-07	1.00E-05	NA
Arsenic	1.40E-05	2.61E-06	1.70E-07	4.43E-06	3.21E-05	NA
Mercury	2.01E-07	NA	NA	NA	NA	8.40E-06
MeHg	8.09E-10	6.04E-07	5.48E-08	9.42E-06	4.70E-07	NA
COPC	**Oral - Soil LADD**	**Oral - Beaver LADD**	**Oral - Mallard Duck LADD**	**Oral - Pike LADD**	**Oral - Berries LADD**	**Inhalation EC**
Arsenic	4.19E-06	7.84E-07	5.09E-08	1.33E-06	9.64E-06	NA

COPC = constituent of potential concern
ADD = average daily dose (mg/kg BW/day)
EC = effect concentration (µg/m³)
LADD = lifetime average daily dose (mg/kg BW/day)
NA = Not applicable (mercury in biota assumed to be 100% MeHg; antimony and arsenic are negligible in baseline air; MeHg is not present in air)

TABLE 5.6-2: BASELINE DOSE OF CONSTITUENTS OF POTENTIAL CONCERN FOR CHILD SUBSISTENCE

COPC	Oral – Soil ADD	Oral - Beaver ADD	Oral - Mallard Duck ADD	Oral - Pike ADD	Oral - Berries ADD	Inhalation EC
Antimony	4.86E-05	4.42E-05	4.01E-06	2.41E-06	3.48E-05	NA
Arsenic	1.49E-04	9.10E-06	5.90E-07	1.54E-05	1.12E-04	NA
Mercury	2.14E-06	NA	NA	NA	NA	8.40E-06
MeHg	8.63E-09	2.10E-06	1.91E-07	3.28E-05	1.64E-06	NA
COPC	**Oral – Soil LADD**	**Oral - Beaver LADD**	**Oral - Mallard Duck LADD**	**Oral - Pike LADD**	**Oral - Berries LADD**	**Inhalation EC**
Arsenic	1.28E-05	7.80E-07	5.06E-08	1.32E-06	9.59E-06	NA

COPC = constituent of potential concern
ADD = average daily dose (mg/kg BW/day)
EC = effect concentration (µg/m³)
NA = Not applicable (mercury in biota assumed to be 100% MeHg; antimony and arsenic are negligible in baseline air; MeHg is not present in air)

TABLE 5.6-3: FUTURE DOSE OF CONSTITUENTS OF POTENTIAL CONCERN FOR ADULT SUBSISTENCE

COPC	Oral - Soil ADD	Oral - Beaver ADD	Oral - Mallard Duck ADD	Oral - Pike ADD	Oral – Berries ADD	Inhalation EC
Antimony	4.56E-06	1.27E-05	1.15E-06	6.94E-07	1.00E-05	5.71E-07
Arsenic	1.41E-05	2.64E-06	1.71E-07	4.44E-06	3.25E-05	1.80E-05
Mercury	2.03E-07	NA	NA	NA	NA	8.46E-06
MeHg	8.49E-10	6.10E-07	5.54E-08	9.89E-06	4.94E-07	NA
COPC	**Oral - Soil LADD**	**Oral - Beaver LADD**	**Oral - Mallard Duck LADD**	**Oral - Pike LADD**	**Oral – Berries LADD**	**Inhalation EC**
Arsenic	4.24E-06	7.91E-07	5.13E-08	1.33E-06	9.75E-06	5.39E-06

COPC = constituent of potential concern
ADD = average daily dose (mg/kg BW/day)
EC = effect concentration (µg/m³)
LADD = lifetime average daily dose (mg/kg BW/day)
NA = Not applicable (mercury in biota assumed to be 100% MeHg; MeHg not present in air)

TABLE 5.6-4: FUTURE DOSE OF CONSTITUENTS OF POTENTIAL CONCERN FOR CHILD SUBSISTENCE

COPC	Oral - Soil ADD	Oral - Beaver ADD	Oral - Mallard Duck ADD	Oral - Pike ADD	Oral – Berries ADD	Inhalation EC
Antimony	4.87E-05	4.43E-05	4.02E-06	2.42E-06	3.48E-05	5.71E-07
Arsenic	1.51E-04	9.18E-06	5.96E-07	1.55E-05	1.13E-04	1.80E-05
Mercury	2.16E-06	NA	NA	NA	NA	8.46E-06
MeHg	9.06E-09	2.12E-06	1.93E-07	3.44E-05	1.72E-06	NA
COPC	**Oral - Soil LADD**	**Oral - Beaver LADD**	**Oral - Mallard Duck LADD**	**Oral - Pike LADD**	**Oral – Berries LADD**	**Inhalation EC**
Arsenic	1.29E-05	7.87E-07	5.11E-08	1.33E-06	9.70E-06	1.54E-06

COPC = constituent of potential concern
ADD = average daily dose (mg/kg BW/day)
EC = effect concentration (µg/m³)
NA = Not applicable (mercury in biota assumed to be 100% MeHg; MeHg not present in air)